

6. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. This chapter provides an assessment of non-carbon dioxide emissions from the following source categories: enteric fermentation in domestic livestock, livestock manure management, rice cultivation, agricultural soil management, and field burning of agricultural residues (see Figure 6-1). Carbon dioxide (CO₂) emissions and removals from agriculture-related land-use activities, such as conversion of grassland to cultivated land, are presented in the Land-Use Change and Forestry sector. Carbon dioxide emissions from on-farm energy use are accounted in the Energy chapter.

Figure 6-1: 2003 Agriculture Chapter Greenhouse Gas Emission Sources

In 2003, the agricultural sector was responsible for emissions of 433.3 Tg CO₂ Eq., or 6.3 percent of total U.S. greenhouse gas emissions. Methane (CH₄) and nitrous oxide (N₂O) were the primary greenhouse gases emitted by agricultural activities. Methane emissions from enteric fermentation and manure management represent about 21 percent and 7 percent of total CH₄ emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were by far the largest emitters of CH₄. Rice cultivation and agricultural crop residue burning were minor sources of CH₄. Agricultural soil management activities such as fertilizer application and other cropping practices were the largest source of U.S. N₂O emissions, accounting for 67 percent. Manure management and field burning of agricultural residues were also small sources of N₂O emissions.

Table 6-1 and Table 6-2 present emission estimates for the Agriculture sector. Between 1990 and 2003, CH₄ emissions from agricultural activities increased by 3.2 percent while N₂O emissions increased by 0.7 percent. In addition to CH₄ and N₂O, field burning of agricultural residues was also a minor source of the ambient air pollutants carbon monoxide (CO) and nitrogen oxides (NO_x).

Table 6-1: Emissions from Agriculture (Tg CO₂ Eq.)

Gas/Source	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	156.9	163.0	164.2	164.6	162.0	161.9	161.5	161.8
Enteric Fermentation	117.9	118.3	116.7	116.8	115.6	114.5	114.6	115.0
Manure Management	31.2	36.4	38.8	38.8	38.1	38.9	39.3	39.1
Rice Cultivation	7.1	7.5	7.9	8.3	7.5	7.6	6.8	6.9
Field Burning of Agricultural Residues	0.7	0.8	0.8	0.8	0.8	0.8	0.7	0.8
N₂O	269.6	269.8	285.6	261.3	282.1	275.6	270.9	271.5
Agricultural Soil Management	253.0	252.0	267.7	243.4	263.9	257.1	252.6	253.5
Manure Management	16.3	17.3	17.4	17.4	17.8	18.0	17.9	17.5
Field Burning of Agricultural Residues	0.4	0.4	0.5	0.4	0.5	0.5	0.4	0.4
Total	426.5	432.8	449.8	425.9	444.1	437.5	432.4	433.3

Note: Totals may not sum due to independent rounding.

Table 6-2: Emissions from Agriculture (Gg)

Gas/Source	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	7,470	7,760	7,821	7,838	7,713	7,708	7,689	7,705
Enteric Fermentation	5,612	5,634	5,557	5,561	5,505	5,454	5,458	5,475
Manure Management	1,485	1,733	1,850	1,846	1,813	1,853	1,873	1,864
Rice Cultivation	339	356	376	395	357	364	325	328
Field Burning of Agricultural Residues	33	37	38	37	38	37	34	38
N₂O	870	870	921	843	910	889	874	876

Agricultural Soil Management	816	813	864	785	851	829	815	818
Manure Management	52	56	56	56	57	58	58	57
Field Burning of Agricultural Residues	1	1	1	1	1	1	1	1
CO	689	767	789	767	790	770	706	794
NO_x	28	34	35	34	35	35	33	33

Note: Totals may not sum due to independent rounding.

6.1. Enteric Fermentation (IPCC Source Category 4A)

Methane is produced as part of normal digestive processes in animals. During digestion, microbes resident in an animal's digestive system ferment food consumed by the animal. This microbial fermentation process, referred to as enteric fermentation, produces CH₄ as a by-product, which can be exhaled or eructated by the animal. The amount of CH₄ produced and excreted by an individual animal depends primarily upon the animal's digestive system, and the amount and type of feed it consumes.

Among domesticated animal types, ruminant animals (e.g., cattle, buffalo, sheep, goats, and camels) are the major emitters of CH₄ because of their unique digestive system. Ruminants possess a rumen, or large "fore-stomach," in which microbial fermentation breaks down the feed they consume into products that can be absorbed and metabolized. The microbial fermentation that occurs in the rumen enables them to digest coarse plant material that non-ruminant animals cannot. Ruminant animals, consequently, have the highest CH₄ emissions among all animal types.

Non-ruminant domesticated animals (e.g., swine, horses, and mules) also produce CH₄ emissions through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants emit significantly less CH₄ on a per-animal basis than ruminants because the capacity of the large intestine to produce CH₄ is lower.

In addition to the type of digestive system, an animal's feed quality and feed intake also affect CH₄ emissions. In general, lower feed quality or higher feed intake lead to higher CH₄ emissions. Feed intake is positively related to animal size, growth rate, and production (e.g., milk production, wool growth, pregnancy, or work). Therefore, feed intake varies among animal types as well as among different management practices for individual animal types.

Methane emission estimates from enteric fermentation are provided in Table 6-3 and Table 6-4. Total livestock CH₄ emissions in 2003 were 115 Tg CO₂ Eq. (5,475 Gg), increasing very slightly since 2002 due to minor increases in some animal populations and dairy cow milk production in some regions. Beef cattle remain the largest contributor of CH₄ emissions from enteric fermentation, accounting for 72 percent in 2003. Emissions from dairy cattle in 2003 accounted for 24 percent, and the remaining emissions were from horses, sheep, swine, and goats.

From 1990 to 2003, emissions from enteric fermentation have decreased by 2 percent. Generally, emissions have been decreasing since 1995, mainly due to decreasing populations of both beef and dairy cattle and improved feed quality for feedlot cattle. During this timeframe, populations of sheep and goats have also decreased, while horse populations increased and the populations of swine fluctuated.

Table 6-3: CH₄ Emissions from Enteric Fermentation (Tg CO₂ Eq.)

Livestock Type	1990	1997	1998	1999	2000	2001	2002	2003
Beef Cattle	83.2	86.6	85.0	84.9	83.4	82.4	82.3	82.5
Dairy Cattle	28.9	26.4	26.3	26.6	27.0	26.9	27.1	27.3
Horses	1.9	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Sheep	1.9	1.3	1.3	1.2	1.2	1.2	1.1	1.1
Swine	1.7	1.8	2.0	1.9	1.9	1.9	1.9	1.9
Goats	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Total	117.9	118.3	116.7	116.8	115.6	114.5	114.6	115.0

Note: Totals may not sum due to independent rounding.

Table 6-4: CH₄ Emissions from Enteric Fermentation (Gg)

Livestock Type	1990	1997	1998	1999	2000	2001	2002	2003
Beef Cattle	3,961	4,124	4,047	4,045	3,973	3,923	3,919	3,930
Dairy Cattle	1,375	1,255	1,251	1,265	1,283	1,282	1,290	1,300
Horses	91	93	94	93	94	95	95	95
Sheep	91	64	63	58	56	56	53	50
Swine	81	88	93	90	88	88	90	90
Goats	13	10	10	10	10	10	10	10
Total	5,612	5,634	5,557	5,561	5,505	5,454	5,458	5,475

Note: Totals may not sum due to independent rounding.

Methodology

Livestock emission estimates fall into two categories: cattle and other domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics, account for the majority of CH₄ emissions from livestock in the United States. A more detailed methodology (i.e., IPCC Tier 2) was therefore applied to estimating emissions for all cattle except for bulls. Emission estimates for other domesticated animals (horses, sheep, swine, goats, and bulls) were handled using a less detailed approach (i.e., IPCC Tier 1).

While the large diversity of animal management practices cannot be precisely characterized and evaluated, significant scientific literature exists that describes the quantity of CH₄ produced by individual ruminant animals, particularly cattle. A detailed model that incorporates this information and other analyses of livestock population, feeding practices and production characteristics was used to estimate emissions from cattle populations.

National cattle population statistics were disaggregated into the following cattle sub-populations:

Dairy Cattle

- Calves
- Heifer Replacements
- Cows

Beef Cattle

- Calves
- Heifer Replacements
- Heifer and Steer Stockers
- Animals in Feedlots (Heifers and Steers)
- Cows
- Bulls

Calf birth rates, end-of-year population statistics, detailed feedlot placement information, and slaughter weight data were used to model cohorts of individual animal types and their specific emission profiles. The key variables tracked for each of the cattle population categories are described in Annex 3.9. These variables include performance factors such as pregnancy and lactation, as well as average weights and weight gain. Annual cattle population data were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (1995a,b, 1999a,c,d,f, 2000a,c,d,f, 2001a,c,d,f, 2002a,c,d,f, 2003a,c,d,f, 2004a,c,d,f).

Diet characteristics were estimated by region for U.S. dairy, beef, and feedlot cattle. These estimates were used to calculate Digestible Energy (DE) values and CH₄ conversion rates (Y_m) for each population category. The IPCC recommends Y_m values of 3.5 to 4.5 percent for feedlot cattle and 5.5 to 6.5 percent for other well-fed cattle consuming temperate-climate feed types. Given the availability of detailed diet information for different regions and animal types in the United States, DE and Y_m values unique to the United States were developed, rather than using the recommended IPCC values. The diet characterizations and estimation of DE and Y_m values were based on information from state agricultural extension specialists, a review of published forage quality studies, expert

opinion, and modeling of animal physiology. The diet characteristics for dairy cattle were from Donovan (1999), while beef cattle were derived from NRC (2000). DE and Y_m for dairy cows were calculated from diet characteristics using a model simulating ruminant digestion in growing and/or lactating cattle (Donovan and Baldwin 1999). For feedlot animals, DE and Y_m values recommended by Johnson (1999) were used. Values from EPA (1993) were used for dairy replacement heifers. For grazing beef cattle, DE values were based on diet information in NRC (2000) and Y_m values were based on Johnson (2002). Weight data were estimated from Feedstuffs (1998), Western Dairyman (1998), and expert opinion. See Annex 3.9 for more details on the method used to characterize cattle diets in the United States.

To estimate CH₄ emissions from cattle, the population was divided into region, age, sub-type (e.g., calves, heifer replacements, cows, etc.), and production (i.e., pregnant, lactating, etc.) groupings to more fully capture differences in CH₄ emissions from these animal types. Cattle diet characteristics were used to develop regional emission factors for each sub-category. Tier 2 equations from IPCC (2000) were used to produce CH₄ emission factors for the following cattle types: dairy cows, beef cows, dairy replacements, beef replacements, steer stockers, heifer stockers, steer feedlot animals, and heifer feedlot animals. To estimate emissions from cattle, population data were multiplied by the emission factor for each cattle type. More details are provided in Annex 3.9.

Emission estimates for other animal types were based on average emission factors representative of entire populations of each animal type. Methane emissions from these animals accounted for a minor portion of total CH₄ emissions from livestock in the United States from 1990 through 2003. Also, the variability in emission factors for each of these other animal types (e.g., variability by age, production system, and feeding practice within each animal type) is less than that for cattle. Annual livestock population data for these other livestock types, except horses, as well as feedlot placement information were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a-b, 1995a-b, 1998a-b, 1999a-c, 2000a-g, 2001a-f, 2002a-f, 2003a-f, 2004a-f). Horse population data were obtained from the FAOSTAT database (FAO 2004), because USDA does not estimate U.S. horse populations annually. Goat population data were obtained from the Census of Agriculture (USDA 1999g). Methane emissions from sheep, goats, swine, and horses were estimated by using emission factors utilized in Crutzen et al. (1986, cited in IPCC/UNEP/OECD/IEA 1997). These emission factors are representative of typical animal sizes, feed intakes, and feed characteristics in developed countries. The methodology is the same as that recommended by IPCC (IPCC/UNEP/OECD/IEA 1997, IPCC 2000).

See Annex 3.9 for more detailed information on the methodology and data used to calculate CH₄ emissions from enteric fermentation.

Uncertainty

Uncertainty estimates were developed for the emission estimates presented in EPA (2003). No significant changes occurred in the method of data collection, data estimation methodology, or other factors that influence the uncertainty ranges around the 2003 activity data and emission factor input variables. Consequently, the EPA (2003) uncertainty estimates were directly applied to the 2003 emission estimates.

A total of 185 primary input variables (178 for cattle and 8 for non-cattle) were identified as key input variables for the uncertainty analysis. A normal distribution was assumed for almost all activity- and emission factor-related input variables. A triangular distribution was assigned for three input variables (specifically cow-birth ratios for the current and the past two years). For some key input variables, the uncertainty ranges around their estimates (used for inventory estimation) were collected from published documents and other public sources. In addition, both endogenous and exogenous correlations between selected primary input variables were modeled. The exogenous correlation coefficients between the probability distributions of selected activity-related variables were developed as educated estimates.

The uncertainty ranges associated with the activity-related input variables were no larger in magnitude than plus or minus 10 percent. However, for many emission factor-related input variables, the lower- and/or upper-bound uncertainty estimates were over 20 percent. The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 6-5. Enteric fermentation CH₄ emissions in 2003 were estimated to be between 102.3 and 135.7 Tg CO₂ Eq. at a 95 percent confidence level (or in 19 out of 20 Monte Carlo Stochastic Simulations). This indicates a range of 11 percent below to 18 percent above the 2003 emission estimate of 115.0 Tg CO₂ Eq. Among

the individual sub-source categories, beef cattle accounts for the largest amount of CH₄ emissions as well as the largest degree of uncertainty in the emission estimates. Consequently, the cattle sub-source categories together contribute to the largest degree of uncertainty to the estimates of CH₄ emissions from livestock enteric fermentation. Among non-cattle, horses account for the largest degree of uncertainty in the emission estimates.

Table 6-5: Tier 2 Quantitative Uncertainty Estimates for CH₄ Emissions from Enteric Fermentation (Tg CO₂ Eq. and Percent)

Source	Gas	2003 Emission Estimate (Tg CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
			(Tg CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Enteric Fermentation	CH ₄	115.0	102.3	135.7	-11%	+18%

^a Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

QA/QC and Verification

In order to ensure the quality of the emission estimates from enteric fermentation, the IPCC Tier 1 and Tier 2 Quality Assurance/Quality Control (QA/QC) procedures were implemented consistent with the U.S. QA/QC plan. Tier 2 QA procedures included independent peer review of emission estimates. Particular emphasis was placed on cattle population and growth data, and on evaluating the effects of data updates as described in the recalculations discussion below.

Recalculations Discussion

While there were no changes in the methodologies used for estimating CH₄ emissions from enteric fermentation, emissions were revised slightly due to changes in historical data. USDA published revised population estimates in 2004 for some cattle statistics; these include population, livestock placements, and slaughter statistics for 2000, 2001, and 2002. Emission estimates changed for these years for both beef and dairy cattle as a result of revised inputs that reflect USDA updates.

The rate of weight gain for growing steers and heifers was increased for the modeling of 2000 through 2003. The model uses the weight gain data to estimate the number of cattle (steers and heifers) available to be placed into feedlots (by weight class). These estimates were compared to the USDA statistics on actual feedlot placements (by weight class). The updated USDA data show increases in feedlot placements in the heavy weight classes, and required an increase in the rate of weight gain in the modeled population in order to match the observed statistics. Additionally, the distribution of cattle by weight at the start of the year was adjusted to reflect the larger portion of heavier animals.

In 2000, both beef and dairy cattle emissions changed less than 3 Gg (0.1 percent) as a result of the recalculations. In 2001, beef cattle CH₄ emissions increased 12 Gg (0.3 percent), while dairy cattle emissions decreased 1 Gg (0.1 percent). In 2002, beef cattle CH₄ emissions increased 8 Gg (0.2 percent), while dairy cattle emissions increased less than 1 Gg (0.03 percent). For other livestock types, a slight upward revision in the swine population for 2002 resulted in an increase in CH₄ emissions of less than 1 Gg (0.06 percent) in that year. Overall, the changes resulted in an average annual increase of less than 0.1 Tg CO₂ Eq. (0.04 percent) in CH₄ emissions from enteric fermentation for the period 1990 through 2002.

Planned Improvements

The revised and updated USDA data discussed above highlight the need to re-examine several model inputs. Although the enteric fermentation model was constructed to identify the imbalances mentioned in the recalculations discussion, the current inventory presents the first effort to address such differences by making adjustments to model inputs. The updates are based both on expert opinion and on equations published by the American Society of Agricultural Engineers (ASAE) that predict weight versus age statistics for steers and imply growth rates larger than

those used in previous years (ASAE 1999). In addition, in 2001, USDA reported increased rates of gains for yearlings (USDA 2001g). While these two sources provide support for the updates, further research is necessary to verify the changes and to understand what changes over time may be necessary in future inventory analyses.

6.2. Manure Management (IPCC Source Category 4B)

The management of livestock manure can produce anthropogenic CH₄ and N₂O emissions. Methane is produced by the anaerobic decomposition of manure. Nitrous oxide is produced as part of the nitrogen cycle through the nitrification and denitrification of the organic nitrogen in livestock manure and urine.

When livestock or poultry manure are stored or treated in systems that promote anaerobic conditions (e.g., as a liquid/slurry in lagoons, ponds, tanks, or pits), the decomposition of materials in the manure tends to produce CH₄. When manure is handled as a solid (e.g., in stacks or pits) or deposited on pasture, range, or paddock lands, it tends to decompose aerobically and produce little or no CH₄. A number of other factors related to how the manure is handled also affect the amount of CH₄ produced. Ambient temperature, moisture, and manure storage or residency time affect the amount of CH₄ produced because they influence the growth of the bacteria responsible for CH₄ formation. For example, CH₄ production generally increases with rising temperature and residency time. Also, for non-liquid-based manure systems, moist conditions (which are a function of rainfall and humidity) favor CH₄ production. Although the majority of manure is handled as a solid, producing little CH₄, the general trend in manure management, particularly for large dairy and swine producers, is one of increasing use of liquid systems. In addition, use of daily spread systems at smaller dairies is decreasing, due to new regulations limiting the application of manure nutrients, which has resulted in an increase of manure managed and stored on site at these smaller dairies.

The composition of the manure also affects the amount of CH₄ produced. Manure composition varies by animal type, including the animal's digestive system and diet. In general, the greater the energy content of the feed, the greater the potential for CH₄ emissions. For example, feedlot cattle fed a high-energy grain diet generate manure with a high CH₄-producing capacity. Range cattle fed a low energy diet of forage material produce manure with about 50 percent of the CH₄-producing potential of feedlot cattle manure. However, some higher energy feeds also are more digestible than lower quality forages, which can result in less overall waste excreted from the animal. Ultimately, a combination of diet types and the growth rate of the animals will affect the quantity and characteristics of the manure produced.

A very small portion of the total nitrogen excreted is expected to convert to N₂O in the waste management system. The production of N₂O from livestock manure depends on the composition of the manure and urine, the type of bacteria involved in the process, and the amount of oxygen and liquid in the manure system. For N₂O emissions to occur, the manure must first be handled aerobically where ammonia or organic nitrogen is converted to nitrates and nitrites (nitrification), and then handled anaerobically where the nitrates and nitrites are reduced to nitrogen gas (N₂), with intermediate production of N₂O and nitric oxide (NO) (denitrification) (Groffman et al. 2000). These emissions are most likely to occur in dry manure handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to saturation. For example, manure at cattle drylots is deposited on soil, oxidized to nitrite and nitrate, and has the potential to encounter saturated conditions following rain events.

Certain N₂O emissions are accounted for and discussed in the Agricultural Soil Management source category within the Agriculture sector. These are emissions from livestock manure and urine deposited on pasture, range, or paddock lands, as well as emissions from manure and urine that is spread onto fields either directly as "daily spread" or after it is removed from manure management systems (e.g., lagoon, pit, etc.).

Table 6-6 and Table 6-7 provide estimates of CH₄ and N₂O emissions from manure management by animal category. Estimates for CH₄ emissions in 2003 were 39.1 Tg CO₂ Eq. (1,864 Gg), 25 percent higher than in 1990. The majority of this increase was from swine and dairy cow manure, where emissions increased 30 and 38 percent, respectively. The increase in emissions from these animal types is primarily attributed to shifts by the swine and dairy industries towards larger facilities. Larger swine and dairy farms tend to use liquid systems to manage (flush or scrape) and store manure. Thus the shift toward larger facilities is translated into an increasing use of liquid manure management systems, which have higher potential CH₄ emissions than dry systems. This shift was

accounted for by incorporating state-specific weighted CH₄ conversion factor (MCF) values in combination with the 1992 and 1997 farm-size distribution data reported in the *Census of Agriculture* (USDA 1999e). From 2002 to 2003, there was a 0.5 percent decrease in CH₄ emissions, due to minor shifts in the animal populations and the resultant effects on manure management system allocations. A description of the emission estimation methodology is provided in Annex 3.10.

Total N₂O emissions from manure management systems in 2003 were estimated to be 17.5 Tg CO₂ Eq. (57 Gg). The 8 percent increase in N₂O emissions from 1990 to 2003 can be partially attributed to a shift in the poultry industry away from the use of liquid manure management systems, in favor of litter-based systems and high-rise houses. In addition, there was an overall increase in the population of poultry and swine from 1990 to 2002, although swine populations periodically declined slightly throughout the time series. Nitrous oxide emissions showed a 2 percent decrease from 2002 to 2003, due to minor shifts in animal population.

The population of beef cattle in feedlots increased over the period of 1990 to 2003, resulting in increased N₂O emissions from this sub-category of cattle. Although dairy cow populations decreased overall for the period 1990 to 2003, the population of dairies managing and storing manure on-site—as opposed to using pasture, range, or paddock or daily spread systems—increased. Over the same period, dairies also experienced a shift to more liquid manure management systems at large operations, which result in lower N₂O emissions than dry systems. The net result is a slight decrease in dairy cattle N₂O emissions over the period 1990 to 2003. As stated previously, N₂O emissions from livestock manure deposited on pasture, range, or paddock land and manure immediately applied to land in daily spread systems are accounted for in the Agricultural Soil Management source category of the Agriculture sector.

Table 6-6: CH₄ and N₂O Emissions from Manure Management (Tg CO₂ Eq.)

Gas/Animal Type	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	31.2	36.4	38.8	38.8	38.1	38.9	39.3	39.1
Dairy Cattle	11.4	13.4	13.9	14.7	14.5	15.0	15.2	15.7
Beef Cattle	3.2	3.2	3.1	3.1	3.1	3.1	3.1	3.1
Swine	13.1	16.4	18.4	17.6	17.1	17.4	17.7	17.0
Sheep	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Goats	+	+	+	+	+	+	+	+
Poultry	2.7	2.7	2.7	2.6	2.6	2.7	2.7	2.7
Horses	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
N₂O	16.3	17.3	17.4	17.4	17.8	18.0	17.9	17.5
Dairy Cattle	4.3	4.0	3.9	4.0	4.0	3.9	3.9	3.9
Beef Cattle	4.9	5.4	5.5	5.5	5.9	6.1	5.9	5.6
Swine	0.4	0.4	0.5	0.4	0.4	0.4	0.4	0.4
Sheep	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Goats	+	+	+	+	+	+	+	+
Poultry	6.4	7.2	7.2	7.2	7.2	7.3	7.4	7.3
Horses	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Total	47.4	53.7	56.2	56.2	55.9	57.0	57.3	56.7

+ Does not exceed 0.05 Tg CO₂ Eq.

Note: Totals may not sum due to independent rounding.

Table 6-7: CH₄ and N₂O Emissions from Manure Management (Gg)

Gas/Animal Type	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	1,485	1,733	1,850	1,846	1,813	1,853	1,873	1,864
Dairy Cattle	545	639	662	700	692	715	722	748
Beef Cattle	153	152	149	150	149	148	147	146
Swine	622	780	874	837	812	826	843	808
Sheep	9	6	6	6	5	5	5	5

Goats	1	1	1	1	1	1	1	1
Poultry	128	127	130	125	125	129	126	127
Horses	27	28	28	28	28	29	29	29
N₂O	52	56	56	56	57	58	58	57
Dairy Cattle	14	13	13	13	13	13	13	13
Beef Cattle	16	17	18	18	19	20	19	18
Swine	1	1	1	1	1	1	1	1
Sheep	+	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+	+
Poultry	21	23	23	23	23	24	24	24
Horses	1	1	1	1	1	1	1	1

+ Does not exceed 0.5 Gg.

Note: Totals may not sum due to independent rounding.

Methodology

The methodologies presented in *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000) form the basis of the CH₄ and N₂O emission estimates for each animal type. The calculation of emissions requires the following information:

- Animal population data (by animal type and state);
- Amount of nitrogen produced (excretion rate by animal type times animal population);
- Amount of volatile solids produced (excretion rate by animal type times animal population);
- Methane producing potential of the volatile solids (by animal type);
- Extent to which the CH₄ producing potential is realized for each type of manure management system (by state and manure management system, including the impacts of any biogas collection efforts);
- Portion of manure managed in each manure management system (by state and animal type); and
- Portion of manure deposited on pasture, range, or paddock or used in daily spread systems.

This section presents a summary of the methodologies used to estimate CH₄ and N₂O emissions from manure management for this inventory. See Annex 3.10 for more detailed information on the methodology and data used to calculate CH₄ and N₂O emissions from manure management.

Both CH₄ and N₂O emissions were estimated by first determining activity data, including animal population, waste characteristics, and manure management system usage. For swine and dairy cattle, manure management system usage was determined for different farm size categories using data from USDA (USDA 1996b, 1998d, 2000h) and EPA (ERG 2000a, EPA 2001a, 2001b). For beef cattle and poultry, manure management system usage data was not tied to farm size (ERG 2000a, USDA 2000i, UEP 1999). For other animal types, manure management system usage was based on previous estimates (EPA 1992).

Next, MCFs and N₂O emission factors were determined for all manure management systems. MCFs for dry systems and N₂O emission factors for all systems were set equal to default IPCC factors for temperate climates (IPCC 2000). MCFs for liquid/slurry, anaerobic lagoon, and deep pit systems were calculated based on the forecast performance of biological systems relative to temperature changes as predicted in the van't Hoff-Arrhenius equation (see Annex 3.10 for detailed information on MCF derivations for liquid systems). The MCF calculations model the average monthly ambient temperature, a minimum system temperature, the carryover of volatile solids in the system from month to month due to long storage times exhibited by anaerobic lagoon systems, and a factor to account for management and design practices that result in the loss of volatile solids from lagoon systems.

For each animal group, the base emission factors were then weighted to incorporate the distribution of management systems used within each state and thereby to create an overall state-specific weighted emission factor. To calculate this weighted factor, the percent of manure for each animal group managed in a particular system in a state was multiplied by the emission factor for that system and state, and then summed for all manure management systems in the state.

Methane emissions were estimated using the volatile solids (VS) production for all livestock. For poultry and swine animal groups, for example, volatile solids production was calculated using a national average volatile solids production rate from the *Agricultural Waste Management Field Handbook* (USDA 1996a), which was then multiplied by the average weight of the animal and the state-specific animal population. For most cattle groups, regional animal-specific volatile solids production rates that are related to the diet of the animal for each year of the inventory were used (Lieberman et al., 2004). The resulting volatile solids for each animal group was then multiplied by the maximum CH₄ producing capacity of the waste (B₀) and the state-specific CH₄ conversion factors.

Nitrous oxide emissions were estimated by determining total Kjeldahl nitrogen (TKN)¹ production for all livestock wastes using livestock population data and nitrogen excretion rates based on measurements of excreted manure. For each animal group, TKN production was calculated using a national average nitrogen excretion rate from the *Agricultural Waste Management Field Handbook* (USDA 1996a), which was then multiplied by the average weight of the animal and the state-specific animal population. State-specific weighted N₂O emission factors specific to the type of manure management system were then applied to total nitrogen production to estimate N₂O emissions.

The data used to calculate the inventory estimates were based on a variety of sources. Animal population data for all livestock types, except horses and goats, were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a-b, 1995a-b, 1998a-b, 1999a-c, 2000a-g, 2001a-f, 2002a-f, 2003a-f, 2004a-f). Horse population data were obtained from the FAOSTAT database (FAO 2004), because USDA does not estimate U.S. horse populations annually. Goat population data were obtained from the Census of Agriculture (USDA 1999d). Information regarding poultry turnover (i.e., slaughter) rate was obtained from state Natural Resource Conservation Service (NRCS) personnel (Lange 2000). Dairy cow and swine population data by farm size for each state, used for the weighted MCF and emission factor calculations, were obtained from the *Census of Agriculture*, which is conducted every five years (USDA 1999e).

Manure management system usage data for dairy and swine operations were obtained from USDA's Centers for Epidemiology and Animal Health (USDA 1996b, 1998d, 2000h) for small operations and from preliminary estimates for EPA's Office of Water regulatory effort for large operations (ERG 2000a; EPA 2001a, 2001b). Data for layers were obtained from a voluntary United Egg Producers' survey (UEP 1999), previous EPA estimates (EPA 1992), and USDA's Animal Plant Health Inspection Service (USDA 2000i). Data for beef feedlots were also obtained from EPA's Office of Water (ERG 2000a; EPA 2001a, 2001b). Manure management system usage data for other livestock were taken from previous estimates (EPA 1992). Data regarding the use of daily spread and pasture, range, or paddock systems for dairy cattle were obtained from personal communications with personnel from several organizations, and data provided by those personnel (Poe et al. 1999). These organizations include state NRCS offices, state extension services, state universities, USDA National Agriculture Statistics Service (NASS), and other experts (Deal 2000, Johnson 2000, Miller 2000, Stettler 2000, Sweeten 2000, and Wright 2000). Additional information regarding the percent of beef steer and heifers on feedlots was obtained from contacts with the national USDA office (Milton 2000).

Methane conversion factors for liquid systems were calculated based on average ambient temperatures of the counties in which animal populations were located. The average county and state temperature data were obtained from the National Climate Data Center (NOAA 2004), and the county population data were calculated from state-level population data from NASS and county-state distribution data from the 1992 and 1997 Census data (USDA 1999e). County population distribution data for 1990 and 1991 were assumed to be the same as 1992; county

¹ Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

population distribution data for 1998 through 2003 were assumed to be the same as 1997; and county population distribution data for 1993 through 1996 were extrapolated based on 1992 and 1997 data.

The maximum CH₄ producing capacity of the volatile solids, or B₀, was determined based on data collected in a literature review (ERG 2000b). B₀ data were collected for each animal type for which emissions were estimated.

Nitrogen excretion rate data from the *USDA Agricultural Waste Management Field Handbook* (USDA 1996a) were used for all livestock except sheep, goats, and horses. Data from the American Society of Agricultural Engineers (ASAE 1999) were used for these animal types. Volatile solids excretion rate data from the *USDA Agricultural Waste Management Field Handbook* (USDA 1996a) were used for swine, poultry, bulls, and calves not on feed. In addition, volatile solids production rates from Lieberman et al. (2004) were used for dairy and beef cows, heifers, and steer for each year of the inventory. Nitrous oxide emission factors and MCFs for dry systems were taken from *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000).

Uncertainty

An analysis was conducted for the manure management emission estimates presented in EPA (2003) to determine the uncertainty associated with estimating N₂O and CH₄ emissions from livestock manure management. Because no substantial modifications were made to the inventory methodology since the development of these estimates, it is expected that this analysis is applicable to the uncertainty associated with the current manure management emission estimates.

The EPA (2003) quantitative uncertainty analysis for this source category was performed through the IPCC-recommended Tier 2 uncertainty estimation methodology, Monte Carlo Stochastic Simulation technique. The uncertainty analysis was developed based on the methods used to estimate N₂O and CH₄ emissions from manure management systems. A normal probability distribution was assumed for each source data category. The series of equations used were condensed into a single equation for each animal type and state. The equations for each animal group contained four to five variables around which the uncertainty analysis was performed for each state.

The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 6-8. Manure management CH₄ emissions in 2003 were estimated to be between 32.1 and 47.0 Tg CO₂ Eq. at a 95 percent confidence level (or 19 of 20 Monte Carlo Stochastic Simulations). This indicates a range of 18 percent below to 20 percent above the 2003 emission estimate of 39.1 Tg CO₂ Eq. At the 95 percent confidence level, N₂O emissions were estimated to be between 14.7 and 21.7 Tg CO₂ Eq. (or approximately 16 percent below and 24 percent above the 2003 emission estimate of 17.5 Tg CO₂ Eq.).

Table 6-8: Tier 2 Quantitative Uncertainty Estimates for CH₄ and N₂O Emissions from Manure Management (Tg CO₂ Eq. and Percent)

Source	Gas	2003 Emission Estimate (Tg CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
			(Tg CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Manure Management	CH ₄	39.1	32.1	47.0	-18%	+20%
Manure Management	N ₂ O	17.5	14.7	21.7	-16%	+24%

^aRange of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

The primary factors that contribute to the uncertainty in emission estimates are a lack of information on the usage of various manure management systems in each regional location and the exact CH₄ generating characteristics of each type of manure management system. Because of significant shifts in the swine and dairy sectors toward larger farms, it is believed that increasing amounts of manure are being managed in liquid manure management systems. The existing estimates reflect these shifts in the weighted MCFs based on the 1992 and 1997 farm-size data. However, the assumption of a direct relationship between farm size and liquid system usage may not apply in all cases and may vary based on geographic location. In addition, the CH₄ generating characteristics of each manure

management system type are based on relatively few laboratory and field measurements, and may not match the diversity of conditions under which manure is managed nationally.

Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC 2000) published a default range of MCFs for anaerobic lagoon systems of 0 to 100 percent, which reflects the wide range in performance that may be achieved with these systems. There exist relatively few data points on which to determine country-specific MCFs for these systems. In the United States, many livestock waste treatment systems classified as anaerobic lagoons are actually holding ponds that are substantially organically overloaded and therefore not producing CH₄ at the same rate as a properly designed lagoon. In addition, these systems may not be well operated, contributing to higher loading rates when sludge is allowed to enter the treatment portion of the lagoon or the lagoon volume is pumped too low to allow treatment to occur. Rather than setting the MCF for all anaerobic lagoon systems in the United States based on data available from optimized lagoon systems, a MCF methodology was developed that more closely matches observed system performance and accounts for the affect of temperature on system performance.

However, there is uncertainty related to this methodology. The MCF methodology used in the inventory includes a factor to account for management and design practices that result in the loss of volatile solids from the management system. This factor is currently estimated based on data from anaerobic lagoons in temperate climates, and from only three systems. However, this methodology is intended to account for systems across a range of management practices. Future work in gathering measurement data from animal waste lagoon systems across the country will contribute to the verification and refinement of this methodology. It will also be evaluated whether lagoon temperatures differ substantially from ambient temperatures and whether the lower bound estimate of temperature established for lagoons and other liquid systems should be revised for use with this methodology.

The IPCC provides a suggested MCF for poultry waste management operations of 1.5 percent. Additional study is needed in this area to determine if poultry high-rise houses promote sufficient aerobic conditions to warrant a lower MCF.

The default N₂O emission factors published in *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000) were derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce CH₄ at different rates, and would in all likelihood produce N₂O at different rates, although a single N₂O emission factor was used for both system types. In addition, there are little data available to determine the extent to which nitrification-denitrification occurs in animal waste management systems. Ammonia concentrations that are present in poultry and swine systems suggest that N₂O emissions from these systems may be lower than predicted by the IPCC default factors. At this time, there are insufficient data available to develop U.S.-specific N₂O emission factors; however, this is an area of on-going research, and warrants further study as more data become available.

Uncertainty also exists with the maximum CH₄ producing potential of volatile solids excreted by different animal groups (i.e., B₀). The B₀ values used in the CH₄ calculations are published values for U.S. animal waste. However, there are several studies that provide a range of B₀ values for certain animals, including dairy and swine. The B₀ values chosen for dairy assign separate values for dairy cows and dairy heifers to better represent the feeding regimens of these animal groups. For example, dairy heifers do not receive an abundance of high energy feed and consequently, dairy heifer manure will not produce as much CH₄ as manure from a milking cow. However, the data available for B₀ values are sparse, and do not necessarily reflect the rapid changes that have occurred in this industry with respect to feed regimens.

QA/QC and Verification

Tier 1 and Tier 2 QA/QC activities were conducted consistent with the U.S. QA/QC plan. Tier 2 activities focused on comparing estimates for the 2002 and 2003 Inventories for N₂O emissions from managed systems and CH₄ emissions from livestock manure. All errors identified were corrected. Order of magnitude checks were also conducted, and corrections made where needed. Manure nitrogen data were quality assured by comparing state-level data with bottom up estimates derived at the county level and summed to the state level. Similarly, a comparison

was made by animal and waste management system type for the full time series, between national level estimates for nitrogen excreted and the sum of county estimates for the full time series. Efforts also continue to transition various components of the manure management inventory into a database to facilitate current and future QA checks.

Recalculations Discussion

No changes have been incorporated into the overall methodology for the manure management emission estimates; however, changes were made to the calculation of CH₄ emissions from sheep, goats, and horses. Changes were also made to address errors and updates in the population and waste management system data from previous inventory submittals. Additionally the population distribution of horses and poultry were adjusted, the typical animal mass for sheep was adjusted, and the temperature estimations were changed to reflect a refined methodology. Each of these changes is described in detail below.

- Methane emission estimation from sheep, goats, and horses. The sheep, goats, and horses emission methodologies were changed to be consistent with the methodologies used for the other animal groups. Previously, the sheep, goat, and horse methane estimates were scaled based on population data and earlier estimates of methane emissions (EPA 1992).
- Population. All USDA data from 1998 through the present year underwent review pursuant to USDA NASS annual review procedures. The population data in these years reflect some adjustments due to this review. For horses, state-level populations were estimated using the national FAO population data and the state distributions from the 1992 and 1997 Census of Agriculture. For poultry, populations for states reporting non-disclosed populations were estimated by distributing population values attributed to “other” states.
- Waste management system. The waste management system data for poultry were adjusted based on more recent data. Previously, layers were estimated to be 99 percent managed (EPA 1992). More recent WMS data available from USDA's Animal Plant and Health Inspection Service Layers '99 study (USDA 2000i) and the United Egg Producers Study (UEP 1999) indicate that layers are 100 percent managed. Therefore, the layer WMS estimates have been updated accordingly. Also, the waste management system distribution for dairy cows was adjusted to correct rounding errors.
- Typical animal mass. The typical animal mass for sheep were reevaluated and adjusted. Typical animal mass of sheep was adjusted from 27 kg to 68.6 kg (see Annex 3.10 for details).
- Temperature data: Temperature data are not available for every county with animal populations. Previously, counties without temperature data were not accounted for in the estimate of average weighted temperature. This methodology was changed to use the state average temperature for counties without temperature data available.

The combination of these changes resulted in an average annual increase of 0.1 Tg CO₂ Eq. (0.3 percent) in CH₄ emissions and an average annual increase of 0.1 Tg CO₂ Eq. (0.4 percent) in N₂O emissions from manure management for the period 1990 through 2002.

Planned Improvements

Currently, temperate zone MCFs are used for non-liquid waste management systems, including pasture, range, and paddock, daily spread, solid storage, and drylot operations. However, there are some states that have an annual average temperature that would fall below 15°C (i.e., “cool”). Therefore, CH₄ emissions from certain non-liquid waste management systems may be overestimated; however, the difference is expected to be relatively small due to the low MCFs for all “dry” management systems. The use of both cool and temperate MCFs for non-liquid waste management systems will be investigated for future inventories.

Although an effort was made to introduce the variability in volatile solids production due to differences in diet for beef and dairy cows, heifers, and steer, further research is needed to confirm and track diet changes over time. A methodology to assess variability in swine volatile solids production would be useful in future inventory estimates.

The American Society of Agricultural Engineers is publishing new standards for manure production characteristics in 2004. These data will be investigated and evaluated for incorporation into future estimates.

The development of the National Ammonia Emissions Inventory for the United States (EPA 2004) used similar data sources to the current estimates of emissions from manure management, and through the course of development of the ammonia inventory, updated waste management distribution data were identified. Future estimates will attempt to reflect these updated data.

The methodology to calculate MCFs for liquid systems will be examined to determine how to account for a maximum temperature in the liquid systems. Additionally, available research will be investigated to develop a relationship between ambient air temperature and temperature in liquid waste management systems in order to improve that relationship in the MCF methodology.

Research will be initiated into the estimation and validation of the maximum CH₄-producing capacity of animal manure (B₀), for the purpose of obtaining more accurate data to develop emission estimates.

The 2002 Census of Agriculture became available in mid-2004. These data will be used to update assumptions that previously relied on the 1992 and 1997 Census of Agriculture.

6.3. Rice Cultivation (IPCC Source Category 4C)

Most of the world's rice, and all rice in the United States, is grown on flooded fields. When fields are flooded, aerobic decomposition of organic material gradually depletes the oxygen present in the soil and floodwater, causing anaerobic conditions in the soil to develop. Once the environment becomes anaerobic, CH₄ is produced through anaerobic decomposition of soil organic matter by methanogenic bacteria. As much as 60 to 90 percent of the CH₄ produced is oxidized by aerobic methanotrophic bacteria in the soil (Holzapfel-Pschorn et al. 1985, Sass et al. 1990). Some of the CH₄ is also leached away as dissolved CH₄ in floodwater that percolates from the field. The remaining un-oxidized CH₄ is transported from the submerged soil to the atmosphere primarily by diffusive transport through the rice plants. Minor amounts of CH₄ also escape from the soil via diffusion and bubbling through floodwaters.

The water management system under which rice is grown is one of the most important factors affecting CH₄ emissions. Upland rice fields are not flooded, and therefore are not believed to produce CH₄. In deepwater rice fields (i.e., fields with flooding depths greater than one meter), the lower stems and roots of the rice plants are dead so the primary CH₄ transport pathway to the atmosphere is blocked. The quantities of CH₄ released from deepwater fields, therefore, are believed to be significantly less than the quantities released from areas with more shallow flooding depths. Some flooded fields are drained periodically during the growing season, either intentionally or accidentally. If water is drained and soils are allowed to dry sufficiently, CH₄ emissions decrease or stop entirely. This is due to soil aeration, which not only causes existing soil CH₄ to oxidize but also inhibits further CH₄ production in soils. All rice in the United States is grown under continuously flooded conditions; none is grown under deepwater conditions. Mid-season drainage does not occur except by accident (e.g., due to levee breach).

Other factors that influence CH₄ emissions from flooded rice fields include fertilization practices (especially the use of organic fertilizers), soil temperature, soil type, rice variety, and cultivation practices (e.g., tillage, seeding and weeding practices). The factors that determine the amount of organic material that is available to decompose (i.e.,

organic fertilizer use, soil type, rice variety,² and cultivation practices) are the most important variables influencing the amount of CH₄ emitted over an entire growing season because the total amount of CH₄ released depends primarily on the amount of organic substrate available. Soil temperature is known to be an important factor regulating the activity of methanogenic bacteria, and therefore the rate of CH₄ production. However, although temperature controls the amount of time it takes to convert a given amount of organic material to CH₄, that time is short relative to a growing season, so the dependence of total emissions over an entire growing season on soil temperature is weak. The application of synthetic fertilizers has also been found to influence CH₄ emissions; in particular, both nitrate and sulfate fertilizers (e.g., ammonium nitrate, and ammonium sulfate) appear to inhibit CH₄ formation.

Rice is cultivated in eight states: Arkansas, California, Florida, Louisiana, Mississippi, Missouri, Oklahoma, and Texas. Soil types, rice varieties, and cultivation practices for rice vary from state to state, and even from farm to farm. However, most rice farmers utilize organic fertilizers in the form of rice residue from the previous crop, which is left standing, disked, or rolled into the fields. Most farmers also apply synthetic fertilizer to their fields, usually urea. Nitrate and sulfate fertilizers are not commonly used in rice cultivation in the United States. In addition, the climatic conditions of Arkansas, southwest Louisiana, Texas, and Florida allow for a second, or ratoon, rice crop. Methane emissions from ratoon crops have been found to be considerably higher than those from the primary crop. This second rice crop is produced from regrowth of the stubble after the first crop has been harvested. Because the first crop's stubble is left behind in ratooned fields, and there is no time delay between cropping seasons (which would allow for the stubble to decay aerobically), the amount of organic material that is available for decomposition is considerably higher than with the first (i.e., primary) crop.

Rice cultivation is a small source of CH₄ in the United States (Table 6-9 and Table 6-10). In 2003, CH₄ emissions from rice cultivation were 6.9 Tg CO₂ Eq. (328 Gg). Although annual emissions fluctuated unevenly between the years 1990 and 2003, ranging from an annual decrease of 11 percent to an annual increase of 17 percent, there was an overall decrease of 3 percent over the thirteen-year period, due to an overall decrease in ratoon crop area.³ The factors that affect the rice acreage in any year vary from state to state, although the price of rice relative to competing crops is the primary controlling variable in most states. Price is the primary factor affecting rice area in Arkansas, as farmers will plant more of what is most lucrative amongst soybeans, rice, and cotton. Government support programs have also been influential by affecting the price received for a rice crop (Slaton 2001b, Mayhew 1997). California rice area is primarily influenced by price and government programs, but is also affected by water availability (Mutters 2001). In Florida, rice acreage is largely a function of the price of rice relative to sugarcane and corn. Most rice in Florida is rotated with sugarcane, but sometimes it is more profitable for farmers to follow their sugarcane crop with sweet corn or more sugarcane instead of rice (Schueneman 1997, 2001b). In Louisiana, rice area is influenced by government support programs, the price of rice relative to cotton, soybeans, and corn, and in some years, weather (Saichuk 1997, Linscombe 2001b). For example, a drought in 2000 caused extensive saltwater intrusion along the Gulf Coast, making over 32,000 hectares unplanted. The dramatic decrease in ratooned area in Louisiana in 2002 was the result of hurricane damage to that state's rice-cropped area. In Mississippi, rice is usually rotated with soybeans, but if soybean prices increase relative to rice prices, then some of the acreage that would have been planted in rice, is instead planted in soybeans (Street 1997, 2001). In Missouri, rice acreage is affected by weather (e.g., rain during the planting season may prevent the planting of rice), the price differential between rice and soybeans or cotton, and government support programs (Stevens 1997, Guethle 2001). In Oklahoma, the state having the smallest harvested rice area, rice acreage is limited to the areas in the state with the right type of land for rice cultivation. Acreage is limited to growers who can afford the equipment, labor, and land for this intensive crop (Lee 2003). Texas rice area is affected mainly by the price of rice, government support programs, and water availability (Klosterboer 1997, 2001b).

Table 6-9: CH₄ Emissions from Rice Cultivation (Tg CO₂ Eq.)

² The roots of rice plants shed organic material, which is referred to as "root exudate." The amount of root exudate produced by a rice plant over a growing season varies among rice varieties.

³ The 11 percent decrease occurred between 1992 and 1993; the 17 percent increase happened between 1993 and 1994.

State	1990	1997	1998	1999	2000	2001	2002	2003
Primary	5.1	5.6	5.8	6.3	5.5	5.9	5.7	5.4
Arkansas	2.1	2.5	2.7	2.9	2.5	2.9	2.7	2.6
California	0.7	0.9	0.8	0.9	1.0	0.8	0.9	0.9
Florida	+	+	+	+	+	+	+	+
Louisiana	1.0	1.0	1.1	1.1	0.9	1.0	1.0	0.8
Mississippi	0.4	0.4	0.5	0.6	0.4	0.5	0.5	0.4
Missouri	0.1	0.2	0.3	0.3	0.3	0.4	0.3	0.3
Oklahoma	+	+	+	+	NA	+	+	+
Texas	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.3
Ratoon	2.1	1.9	2.1	2.0	2.0	1.7	1.1	1.5
Arkansas	+	+	+	+	+	+	+	+
Florida	+	0.1	0.1	0.1	0.1	+	+	+
Louisiana	1.1	1.2	1.2	1.2	1.3	1.1	0.5	1.0
Texas	0.9	0.7	0.8	0.7	0.7	0.6	0.5	0.5
Total	7.1	7.5	7.9	8.3	7.5	7.6	6.8	6.9

+ Less than 0.05 Tg CO₂ Eq.

NA (Not Available)

Note: Totals may not sum due to independent rounding.

Table 6-10: CH₄ Emissions from Rice Cultivation (Gg CH₄)

State	1990	1997	1998	1999	2000	2001	2002	2003
Primary	241	265	279	300	260	283	274	255
Arkansas	102	118	126	138	120	138	128	124
California	34	44	39	43	47	40	45	43
Florida	1	2	2	2	2	1	1	1
Louisiana	46	50	53	52	41	46	45	38
Mississippi	21	20	23	27	19	22	22	20
Missouri	7	10	12	16	14	18	15	15
Oklahoma	+	+	+	+	NA	+	+	+
Texas	30	22	24	22	18	18	18	15
Ratoon	98	91	98	95	97	81	52	73
Arkansas	+	+	+	+	+	+	+	+
Florida	2	3	3	4	2	2	2	2
Louisiana	52	55	59	58	61	52	25	50
Texas	45	33	36	33	34	27	24	22
Total	339	356	376	395	357	364	325	328

+ Less than 0.5 Gg

NA (Not Available)

Note: Totals may not sum due to independent rounding.

Methodology

The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) recommends utilizing harvested rice areas and area-based seasonally integrated emission factors (i.e., amount of CH₄ emitted over a growing season per unit harvested area) to estimate annual CH₄ emissions from rice cultivation. This methodology is followed with the use of U.S.-specific emission factors derived from rice field measurements. Seasonal emissions have been found to be much higher for ratooned crops than for primary crops, so emissions from ratooned and primary areas are estimated separately using emission factors that are representative of the particular growing season. This approach is consistent with *IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000).

The harvested rice areas for the primary and ratoon crops in each state are presented in Table 6-11. Primary crop areas for 1990 through 2003 for all states except Florida and Oklahoma were taken from U.S. Department of Agriculture's *Field Crops Final Estimates 1987-1992* (USDA 1994), *Field Crops Final Estimates 1992-1997*

(USDA 1998), *Crop Production 2000 Summary* (USDA 2001), *Crop Production 2001 Summary* (USDA 2002), *Crop Production 2002 Summary* (USDA 2003), and *Crop Production 2003 Summary* (USDA 2004). Harvested rice areas in Florida, which are not reported by USDA, were obtained from Tom Schueneman (1999b, 1999c, 2000, 2001a) and Arthur Kirstein (2003), Florida agricultural extension agents, Dr. Chris Deren (2002) of the Everglades Research and Education Centre at the University of Florida, and Gaston Cantens (2004), Vice President of Corporate Relations of the Florida Crystals Company. Harvested rice areas for Oklahoma, which also are not reported by USDA, were obtained from Danny Lee of the Oklahoma Farm Services Agency (Lee 2003, 2004). Acreages for the ratoon crops were derived from conversations with the agricultural extension agents in each state. In Arkansas, ratooning occurred only in 1998 and 1999, when the ratooned area was less than 1 percent of the primary area (Slaton 1999, 2000, 2001a). In Florida, the ratooned area was 50 percent of the primary area from 1990 to 1998 (Schueneman 1999a), about 65 percent of the primary area in 1999 (Schueneman 2000), around 41 percent of the primary area in 2000 (Schueneman 2001a), about 60 percent of the primary area in 2001 (Deren 2002), about 54 percent of the primary area in 2002 (Kirstein 2003) and about 100 percent of the primary area in 2003 (Kirstein 2004). In Louisiana, the percentage of the primary area that was ratooned was constant at 30 percent over the 1990 to 1999 period, increased to approximately 40 percent in 2000, returned to 30 percent in 2001, dropped to 15 percent in 2002, and rose to 35 percent in 2003 (Linscombe 1999a, 2001a, 2002, 2003, 2004 and Bollich 2000). In Texas, the percentage of the primary area that was ratooned was constant at 40 percent over the entire 1990 to 1999 period and in 2001, but increased to 50 percent in 2000 due to an early primary crop; it then decreased to 40 percent in 2001, 37 percent in 2002, and 38 percent in 2003 (Klosterboer 1999, 2000, 2001a, 2002, 2003, Stansel 2004).

Table 6-11: Rice Areas Harvested (Hectares)

State/Crop	1990	1997	1998	1999	2000	2001	2002	2003
Arkansas								
Primary	485,633	562,525	600,971	657,628	570,619	656,010	608,256	588,830
Ratoon*	NO	NO	202	202	NO	NO	NO	NO
California	159,854	208,822	185,350	204,371	221,773	190,611	213,679	205,180
Florida								
Primary	4,978	7,689	8,094	7,229	7,801	4,562	5,077	2,315
Ratoon	2,489	3,845	4,047	4,673	3,193	2,752	2,734	2,315
Louisiana								
Primary	220,558	235,937	250,911	249,292	194,253	220,963	216,512	182,113
Ratoon	66,168	70,781	75,273	74,788	77,701	66,289	32,477	63,739
Mississippi	101,174	96,317	108,458	130,716	88,223	102,388	102,388	94,699
Missouri	32,376	47,349	57,871	74,464	68,393	83,772	73,654	69,203
Oklahoma	617	12	19	220	NA	265	274	53
Texas								
Primary	142,857	104,816	114,529	104,816	86,605	87,414	83,367	72,845
Ratoon	57,143	41,926	45,811	41,926	43,302	34,966	30,846	27,681
Total	1,148,047	1,263,468	1,326,203	1,428,736	1,237,668	1,345,984	1,303,206	1,215,237
Primary								
Total Ratoon	125,799	116,552	125,334	121,589	124,197	104,006	66,056	93,735
Total	1,273,847	1,380,020	1,451,536	1,550,325	1,361,864	1,449,991	1,369,262	1,308,972

* Arkansas ratooning occurred only in 1998 and 1999.

NO (Not Occurring)

NA (Not Available)

Note: Totals may not sum due to independent rounding.

To determine what seasonal CH₄ emission factors should be used for the primary and ratoon crops, CH₄ flux information from rice field measurements in the United States was collected. Experiments which involved atypical or nonrepresentative management practices (e.g., the application of nitrate or sulfate fertilizers, or other substances

believed to suppress CH₄ formation), as well as experiments in which measurements were not made over an entire flooding season or floodwaters were drained mid-season, were excluded from the analysis. The remaining experimental results⁴ were then sorted by season (i.e., primary and ratoon) and type of fertilizer amendment (i.e., no fertilizer added, organic fertilizer added, and synthetic and organic fertilizer added). The experimental results from primary crops with added synthetic and organic fertilizer (Bossio et al. 1999, Cicerone et al. 1992, Sass et al. 1991a and 1991b) were averaged to derive an emission factor for the primary crop, and the experimental results from ratoon crops with added synthetic fertilizer (Lindau and Bollich 1993, Lindau et al. 1995) were averaged to derive an emission factor for the ratoon crop. The resultant emission factor for the primary crop is 210 kg CH₄/hectare-season, and the resultant emission factor for the ratoon crop is 780 kg CH₄/hectare-season.

Uncertainty

The largest uncertainty in the calculation of CH₄ emissions from rice cultivation is associated with the emission factors. Seasonal emissions, derived from field measurements in the United States, vary by more than one order of magnitude. This inherent variability is due to differences in cultivation practices, in particular, fertilizer type, amount, and mode of application; differences in cultivar type; and differences in soil and climatic conditions. A portion of this variability is accounted for by separating primary from ratooned areas. However, even within a cropping season or a given management regime, measured emissions may vary significantly. Of the experiments used to derive the emission factors applied here, primary emissions ranged from 22 to 479 kg CH₄/hectare-season and ratoon emissions ranged from 481 to 1,490 kg CH₄/hectare-season. From these ranges, an uncertainty for the emission factors of 109 percent for primary crops and 65 percent for ratoon was calculated. In order to perform a Tier 2 Monte Carlo uncertainty analysis, some information regarding the statistical distribution of the uncertainty is required. Variability about the rice emission factor means were not normally distributed for either primary or ratooned crops, but rather skewed, with a tail trailing to the right of the mean, therefore a lognormal-type statistical distribution was applied. The bounds of the distribution were set at 0 (indicating that CH₄ absorption was unlikely given this management system) and three times the emission factor.

Uncertainty regarding primary cropping area is an additional consideration. Uncertainty associated with primary rice-cropped area for each state was obtained from expert judgment, and ranged from 1 percent to 5 percent of the mean area. A triangular distribution of uncertainty was assumed about the mean for areas, which was bounded at half and one and a half times the estimated area.

Another source of uncertainty lies in the ratooned areas, which are not compiled regularly. Ratooning accounts for less than 8 percent of the total rice-cropped area, though it is responsible for a proportionately larger portion of emissions. Based on expert judgment, the uncertainty associated with ratooned areas is between 1 percent and 5 percent. A triangular distribution of uncertainty was assumed, and bound at half and one and a half times the estimated proportion of ratooned area.

A final source of uncertainty is in the practice of flooding outside of the normal rice season. According to agricultural extension agents, all of the rice-growing states practice this on some part of their rice acreage. Estimates of these areas range from 5 to 68 percent of the rice acreage. Fields are flooded for a variety of reasons: to provide habitat for waterfowl, to provide ponds for crawfish production, and to aid in rice straw decomposition. To date, however, CH₄ flux measurements have not been undertaken over a sufficient geographic range or under representative conditions to account for this source or its associated uncertainty adequate for inclusion in the emission estimates or uncertainty evaluations presented here.

⁴ In some of these remaining experiments, measurements from individual plots were excluded from the analysis because of the reasons just mentioned. In addition, one measurement from the ratooned fields (i.e., the flux of 2.041 g/m²/day in Lindau and Bollich 1993) was excluded since this emission rate is unusually high compared to other flux measurements in the United States, as well as in Europe and Asia (IPCC/UNEP/OECD/IEA 1997).

To quantify the uncertainties for emissions from rice cultivation, a Monte Carlo (Tier 2) uncertainty analysis was performed using the information provided above. The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 6-12. Rice cultivation CH₄ emissions in 2003 were estimated to be between 2.9 and 13.9 Tg CO₂ Eq. at a 95 percent confidence level (or 19 of 20 Monte Carlo Stochastic Simulations). This indicates a range of 58 percent below to 101 percent above the 2003 emission estimate of 6.9 Tg CO₂ Eq.

Table 6-12: Tier 2 Quantitative Uncertainty Estimates for CH₄ Emissions from Rice Cultivation (Tg CO₂ Eq. and Percent)

Source	Gas	2003 Emission Estimate (Tg CO ₂ Eq.)	Uncertainty Range Relative to Emission Estimate ^a			
			(Tg CO ₂ Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Rice Cultivation	CH ₄	6.9	2.9	13.9	-58%	+101%

^aRange of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

Planned Improvements

In performing a Monte Carlo-type uncertainty analysis, a higher level Tier 2 type emission mean is calculated incidentally. One would expect there to be a difference in the emission means calculated by these different methods, because under the IPCC default Tier 2 method used here to estimate CH₄ emissions, the statistical distribution of all parameters (i.e., activity data and emission factors) is implicitly considered to be normal. As described above, that is not the case with the uncertainty analysis, which allows for several asymmetrical statistical distributions. Here, the lower and upper bounds have been reported, directly from the Monte Carlo analysis. However, the percentages for the upper and lower bounds of the range have been calculated based on the reported emission mean rather than that mean calculated by the Monte Carlo software (as is the case with all reported Tier 2 analyses). Because that mean may represent an improvement to the current Tier 2 methodology, including the higher level Tier 2 estimate in future inventories is being investigated.

6.4. Agricultural Soil Management (IPCC Source Category 4D)

Nitrous oxide is produced naturally in soils through the microbial processes of nitrification and denitrification.⁵ A number of agricultural activities add nitrogen (N) to soils, thereby increasing the amount available for nitrification and denitrification, and ultimately the amount of nitrous oxide (N₂O) emitted. These activities may add N to soils either directly or indirectly (see Figure 6-2). Direct additions occur through various soil management practices and from the deposition of manure on soils by animals on pasture, range, and paddock (PRP) (i.e., by animals whose manure is not managed). Soil management practices that add N to soils include fertilizer use, application of managed livestock manure and sewage sludge, production of N-fixing crops and forages, retention of crop residues, and cultivation of histosols (i.e., soils with a high organic matter content, otherwise known as organic soils).⁶ Only direct emissions from agricultural lands (i.e., croplands and grasslands), along with emissions from PRP manure are included in this section. The direct emissions from forest lands and settlements are presented within the LUCF sector. Indirect nitrous oxide emissions from all land use types resulting from N additions to croplands, grasslands, forestlands, and settlements are also included in this section. These indirect emissions occur through two

⁵ Nitrification and denitrification are two processes within the N cycle that are brought about by certain microorganisms in soils. Nitrification is the aerobic microbial oxidation of ammonium (NH₄) to nitrate (NO₃), and denitrification is the anaerobic microbial reduction of nitrate to N₂. Nitrous oxide is a gaseous intermediate product in the reaction sequence of denitrification, which leaks from microbial cells into the soil and then into the atmosphere. Nitrous oxide is also produced during nitrification, although by a less well understood mechanism (Nevison 2000).

⁶ Cultivation of histosols does not, *per se*, “add” N to soils. Instead, the process of cultivation enhances mineralization of N-rich organic matter that is present in histosols, thereby enhancing N₂O emissions from histosols.

mechanisms: 1) volatilization and subsequent atmospheric deposition of applied N;⁷ and 2) surface runoff and leaching of applied N into groundwater and surface water. Other agricultural soil management activities, such as irrigation, drainage, tillage practices, and fallowing of land, can affect fluxes of N₂O (as well as other greenhouse gases) to and from soils and are partially accounted for in the analysis.

Figure 6-2: Direct N₂O Emissions Pathways from Cropland and Grassland Soils, and Indirect N₂O Emissions Pathways from All Sources.

Agricultural soils are responsible for the majority of U.S. N₂O emissions. Estimated emissions from this source in 2003 were 253.5 Tg CO₂ Eq. (818 Gg N₂O) (see Table 6-13 and Table 6-14). Annual agricultural soil management N₂O emissions fluctuated between 1990 and 2003; however, overall emissions were 0.2% greater in 2003 than in 1990. Year-to-year fluctuations are largely a reflection of annual variations in climate, synthetic fertilizer consumption, and crop production.

Table 6-13: N₂O Emissions from Agricultural Soils (Tg CO₂ Eq.)

	1990	1997	1998	1999	2000	2001	2002	2003
Direct	140.4	155.9	158.6	151.1	156.3	154.5	159.9	155.3
Agricultural Soils	100.1	113.6	116.5	111.0	116.4	113.0	118.5	114.8
Pasture, Range & Paddock Livestock								
Manure	40.2	42.2	42.1	40.1	39.8	41.5	41.4	40.5
Indirect (All Land Use Types)*	112.6	96.2	109.1	92.3	107.6	102.6	92.7	98.2
Total	253.0	252.0	267.7	243.4	263.9	257.1	252.6	253.5

Note: Totals may not sum due to independent rounding.

*Includes cropland, grassland, forest land and settlements.

Table 6-14: N₂O Emissions from Agricultural Soils (Gg)

	1990	1997	1998	1999	2000	2001	2002	2003
Direct	453	503	512	487	504	498	516	501
Agricultural Soils	323	367	376	358	376	365	382	370
Pasture, Range & Paddock Livestock								
Manure	130	136	136	129	129	134	134	131
Indirect (All Land Use Types)*	363	310	352	298	347	331	299	317
Total	816	813	864	785	851	829	815	818

Note: Totals may not sum due to independent rounding.

*Includes cropland, grassland, forest land and settlements.

Estimated direct and indirect N₂O emissions by sub-source category are provided in Table 6-15, Table 6-16, and Table 6-17.

Table 6-15: Direct N₂O Emissions from Agricultural Soils (Tg CO₂ Eq.)

Activity	1990	1997	1998	1999	2000	2001	2002	2003
Mineral Agricultural Soils	97.3	110.8	113.7	108.2	113.6	110.1	115.6	111.9
Histosol Cultivation	2.8	2.9	2.9	2.9	2.9	2.9	2.9	2.9

⁷ These processes entail volatilization of applied N as ammonia (NH₃) and oxides of N (NO_x), transformations of these gases within the atmosphere (or upon deposition), and deposition of the N primarily in the form of particulate ammonium (NH₄), nitric acid (HNO₃), and oxides of N.

Pasture, Range & Paddock Livestock Manure	40.2		42.2	42.1	40.1	39.8	41.5	41.4	40.5
Total	140.4		155.9	158.6	151.1	156.3	154.5	159.9	155.3

Note: Totals may not sum due to independent rounding. Excludes sewage sludge and livestock manure used as commercial fertilizers.

Table 6-16: Direct N₂O Emissions from PRP Livestock Manure (Tg CO₂ Eq.)

Animal Type	1990		1997	1998	1999	2000	2001	2002	2003
Beef Cattle	34.9		37.8	37.6	35.7	35.5	37.1	37.0	36.1
Dairy Cows	1.9		1.4	1.4	1.3	1.3	1.3	1.3	1.3
Swine	0.5		0.2	0.2	0.2	0.2	0.2	0.2	0.2
Sheep	0.4		0.3	0.3	0.3	0.3	0.3	0.2	0.2
Goats	0.2		0.2	0.2	0.2	0.2	0.2	0.2	0.2
Poultry	0.1		0.1	0.1	0.1	0.1	0.1	0.1	0.1
Horses	2.2		2.3	2.3	2.3	2.3	2.3	2.3	2.3
Total	40.2		42.2	42.1	40.1	39.8	41.5	41.4	40.5

Table 6-17: Indirect N₂O Emissions from all Land Use Types* (Tg CO₂ Eq.)

	1990		1997	1998	1999	2000	2001	2002	2003
Volatilization and Atm. Deposition	15.6		16.5	16.4	16.4	16.8	16.4	16.6	16.5
Surface Leaching & Run-Off	97.1		79.6	92.7	75.9	90.8	86.3	76.1	81.8
Total	112.6		96.2	109.1	92.3	107.6	102.6	92.7	98.2

Note: Totals may not sum due to independent rounding.

*Includes cropland, grassland, forest land and settlements.

Methodology

The methodology used to estimate emissions from agricultural soil management is consistent with the Tier 3 approach of the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), as amended by the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC 2000) and *Good Practice Guidance for Land Use, Land-Use Change and Forestry* (IPCC 2003). Current methods divide this N₂O source category into three components: 1) direct emissions from managed soils due to applied N and the cultivation of histosols; 2) direct emissions from soils due to the deposition of manure by livestock on PRP lands; and 3) indirect emissions from soils or water induced by additions of fertilizers, sewage sludge, and livestock manure (both managed and unmanaged) to soils of all land use types.

Annex 3.11 provides more detailed information on the methodologies and data used to calculate N₂O emissions from each of the components.

The methodology applied in this Inventory is a hybrid approach for estimating N₂O emissions from mineral agricultural soils. This involves using the process-based model DAYCENT to estimate emissions from major crops on mineral (i.e., non-histosol) soils, and the IPCC methodology for non-major crops on mineral soils, PRP manure, as well as all emissions from histosols.

Direct N₂O Emissions from Mineral Agricultural Soils

Different methodologies were used in quantifying direct N₂O emissions from mineral agricultural soils with major crop types and those with non-major crop types as described below.

Major Crop Types

The DAYCENT ecosystem model (Del Grosso et al. 2001, Parton et al. 1998) was used to estimate direct soil N₂O emissions from mineral agricultural soils cropped with major crop types. DAYCENT has been parameterized to

simulate most of the major cropping systems (corn, soybean, wheat, alfalfa hay, other hay, sorghum, and cotton) in the United States. These cropping systems simulated by DAYCENT represent approximately 90 percent of total cropped land in the United States. DAYCENT simulates crop growth, soil organic matter decomposition, greenhouse gas fluxes, N deposited by grazing animals, and other biogeochemical processes using daily climate data, land management information, and soil physical properties as model inputs. The scale of DAYCENT simulations is dictated by the scale of available input data. Soil and climate inputs were available for every county with more than 100 acres of agricultural land. Therefore, a single parameter value (e.g., maximum temperature for a particular day) is applied at the county-level for those variables. Land management data (e.g., timing of planting, harvesting, applying fertilizer, intensity of cultivation, rate of fertilizer application) were available at the agricultural region level as defined by the Agricultural Sector Model (McCarl et al. 1993). There are 63 regions in the contiguous United States; most states correspond to one region, except for states that are divided into two or more regions if there is sufficient variability in cropping practices within the state. Although various cropping systems were simulated for each county, the parameters controlling management activities (e.g., when crops were planted/harvested, amount of fertilizer added), did not change within an agricultural region.

Nitrous oxide emissions estimated by DAYCENT account for N additions, crop type, irrigation, and other factors. However, because DAYCENT is a process-based model that simulates the N cycle, N₂O emissions cannot be partitioned into the contribution of N₂O from different N inputs (e.g., N₂O emissions from synthetic fertilizer applications cannot be distinguished from those emissions resulting from manure applications). Therefore, it was not possible to separate out these individual contributors to N₂O flux, as is suggested in the IPCC *Guidelines*.

In addition to simulating N₂O emissions from mineral agricultural soils cropped with major crop types, a DAYCENT simulation was performed of those same areas as though they were covered by native vegetation, so that anthropogenic emissions could be isolated from natural background emissions. Emissions from managed agricultural lands are the result of complex and interactive processes, practices, and inputs arising from anthropogenic intervention. Because removing inputs alone would not reflect the full anthropogenic greenhouse gas signature, managed soil emissions have been compared to those soils under native vegetation as a means of identifying the anthropogenic contribution. The reported estimates of emissions from managed soils therefore represent the difference between simulated emissions from native vegetation and emissions from cropland soils. Estimates of direct N₂O emissions from N applications were based on the total amount of N applied to soils annually through the following practices: 1) the application of synthetic and organic commercial fertilizers, 2) the application of livestock manure through both daily spread operations and through the eventual application of manure that had been stored in manure management systems, 3) the application of sewage sludge, 4) the production of N-fixing crops and forages, and 5) the retention of crop residues (i.e., leaving residues in the field after harvest). For each of these practices, annual N applications were obtained from the following sources:

- Crop-specific N-fertilization rates: Alexander and Smith (1990), Anonymous (1924), Battaglin and Goolsby (1994), Engle and Makela (1947), ERS (1994, 2003), Fraps and Asbury (1931), Ibach and Adams (1967), Ibach et al. (1964), NFA (1946), NRIAI (2003), Ross and Mehring (1938), Skinner (1931), Smalley et al. (1939), Taylor (1994), USDA (1966, 1957, 1954, 1946).
- Manure management information was obtained from Poe et al. (1999), Safley et al. (1992), and personal communications with agricultural experts (Anderson 2000, Deal 2000, Johnson 2000, Miller 2000, Milton 2000, Stettler 2000, Sweeten 2000, Wright 2000). Livestock weight data were obtained from Safely (2000), USDA (1996, 1998d), and ASAE (1999); daily rates of N excretion from ASAE (1999) and USDA (1996). Comparisons of estimates of managed manure production (i.e. non-PRP manure) with estimates of the amount of manure actually consumed by soils showed that manure consumed by soils accounted for approximately one-third of managed manure production). Values for manure consumption (Kellogg et al. 2000; Edmonds et al. 2003) were subtracted from values of managed manure production. Only consumed manure N was applied to agricultural soils. The remainder was assumed to have volatilized during storage and transport. In contrast to the IPCC methodology that only considers volatilization of manure that was applied to soils, the manure that was assumed to volatilize during transport and storage was included in the volatilization component of indirect N₂O emissions. Instead of assuming that 10 percent of synthetic and 20 percent of organic N applied to soils is volatilized and 30 percent of applied N was leached/runoff as

with IPCC methodology, volatilization and N leaching/runoff were internally calculated by the process-based model.

- Sewage sludge: Bastian (2002); USDA (1998a); EPA (1993, 1999); Metcalf and Eddy (1991).
- Nitrogen-fixing crops and forages and retention of crop residue. Using the IPCC approach, these are considered activity data. However, when using DAYCENT, they should not be considered activity data because they are internally generated by the model. In other words, DAYCENT accounts for the influence of N fixation and retention of crop residue on N₂O emissions, but these are not model inputs.
- Historical and modern crop rotation and management information (e.g., timing and type of cultivation, timing of planting/harvest, etc.): Hurd (1930, 1929), Latta (1938), Iowa State College Staff Members (1946), Bogue (1963), Hurt (1994), USDA (2004), USDA (2000h), as extracted by Eve (2001), and revised by Ogle (2002), CTIC (1998), Piper et al. (1924), Hardies and Hume (1927), Holmes (1902, 1929), Spillman (1902, 1905, 1907, 1908), Chilcott (1910), Smith (1911), Kezer ca. (1917), Hargreaves (1993), ERS (2002), Warren (1911), Langtson et al. (1922), Russell et al. (1922), Elliot and Tapp (1928), Elliot (1933), Ellsworth (1929), Garey (1929), Holmes (1929), Hodges et al. (1930), Bonnen and Elliot (1931), Brenner et al. (2002, 2001), Smith et al. (2002).

Applied N was subject to volatilization and leaching/runoff according to the climatic conditions, soil type and condition, crop type, and land management practices such as cultivation and irrigation, as simulated by DAYCENT. These amounts were then applied in the calculation of indirect emissions as described below. The remaining applied soil N was then added to the applied N from N-fixing crops and crop residues to yield total soil N additions for the DAYCENT simulation of direct N₂O emissions from soils cropped with major crop types. Because the model is sensitive to actual interannual variability in those factors to which N₂O emissions are sensitive (e.g., climate), emissions vary through time rather than demonstrate a linear, monotonic response.

Non-Major Crop Types

For lands cropped with non-major crop types, the IPCC emission factor methodology was used to estimate N₂O emissions from mineral agricultural soils, as described below.

Estimates of direct N₂O emissions from N applications to non-major crop types were based on the amount of N applied to soils annually through the following practices: 1) the application of synthetic commercial fertilizers, 2) the production of N-fixing crops and forages, and 3) the retention of crop residues. No organics were considered here because 100 percent of these were assumed to be applied to crops simulated by DAYCENT. This assumption is reasonable because DAYCENT simulated the 6 major cropping systems (corn, hay, pasture, sorghum, soybean, wheat) that receive the vast majority (approximately 95 percent) of manure applications (Kellogg et al. 2000, Edmonds et al. 2003).

Yearly synthetic fertilizer N additions to non-major crop types were calculated by process of elimination. For each year, fertilizer accounted for by the cropping systems simulated by DAYCENT (approximately 75 percent of the U.S. total), fertilizer estimated to be applied to forests (less than 1 percent of the U.S. total), and fertilizer estimated to be applied in settlements (approximately 10 percent of the U.S. total) were summed and subtracted from total fertilizer used in the United States. This difference was assumed to be applied to non-major crop types and accounted for approximately 15 percent of total N fertilizer used in the United States. Non-major crop types include fruits, nuts, and vegetables, which account for approximately 5 percent of U.S. N fertilizer use (TFI 2000) and other crops not simulated by DAYCENT (barley, oats, tobacco, sugar cane, sugar beets, sunflower, millet, peanuts, etc.) which account for approximately 10 percent of total U.S. fertilizer use. The non-volatilized proportion was obtained by reducing total applications by the default IPCC volatilization fraction (IPCC 1997, 2000). In addition to synthetic fertilizer-N applied to non-major crop types, N in soils due to the cultivation of non-major N-fixing crops (e.g., edible legumes) was included in these estimates. Finally, crop residue N retention was derived from information about which residues are typically left on the field, the fractions that remain, annual crop production,

mass ratios of aboveground residue to crop product, and dry matter fractions and N contents of the residues. For each of these practices, annual N applications were obtained from the following sources:

- Mass ratios of aboveground residue to crop product, dry matter fractions, and N contents for N-fixing crops: Strehler and Stützel (1987), Barnard and Kristoferson (1985), Karkosh (2000), Ketzi (1999), IPCC/UNEP/OECD/IEA (1997).
- Annual production statistics for crops whose residues are left on the field: USDA (1994a, 1998b, 2000i, 2001a, 2002a, 2003a), Schueneman (1999, 2001), Deren (2002), Schueneman and Deren (2002), Cantens (2004), Lee (2003, 2004).
- Aboveground residue to crop mass ratios, residue dry matter fractions, and residue N contents: Strehler and Stützel (1987), Turn et al. (1997), Ketzi (1999), Barnard and Kristoferson (1985), Karkosh (2000).

The net amount of N remaining on the soil from applied fertilizer was added to the N from N-fixing crops and crop residues to yield total unvolatilized applied N, which was multiplied by the IPCC default emission factor to derive an estimate of cropland N₂O emissions from non-major crop types.

Total annual emissions from major crops and other crops were summed to obtain total emissions from cropped mineral soils (see Table 6-13 and Table 6-14).

Direct N₂O Emissions from Histosols

Estimates of annual N₂O emissions from histosol cultivation were based on estimates of the total U.S. acreage of histosols cultivated annually for each of two climatic zones: 1) temperate, and 2) sub-tropical. Histosol area was obtained from the Natural Resources Inventory (USDA 2000h, as extracted by Eve 2001, and revised by Ogle 2002). To estimate annual emissions, the total temperate area was multiplied by the IPCC default emission factor for temperate regions, and the total sub-tropical area was multiplied by the average of the IPCC default emission factors for temperate and tropical regions.

Total Direct N₂O Emissions from Nitrogen Applications to Agricultural Soils

Total annual N₂O emissions from N applications to mineral agricultural soils and annual N₂O emissions from histosol cultivation were then summed to estimate total direct N₂O emissions from agricultural soils.

Direct N₂O Emissions from Pasture, Range, and Paddock Livestock Manure

As with N₂O from major row crops, dual methodologies incorporating the process-based simulation model DAYCENT and IPCC methods were applied in tandem to estimate total emissions from PRP manure. For DAYCENT simulations, annual county-level pasture area data were not available so county-level pasture area estimates from Kellogg et al. (2000) and Edmonds et al. (2003) were used. DAYCENT does not simulate paddocks and no county level area data for rangeland were available so IPCC methodology was used to estimate emissions from these sources. Because DAYCENT simulated only pastures and not paddocks or rangeland, the amount of manure accounted for by DAYCENT (manure N added to soil is an output variable in DAYCENT) was subtracted from annual estimates of total PRP manure and assumed that this manure contributed to emissions from paddocks and rangeland.

Estimates of N₂O emissions from PRP livestock manure are based on the amount of N in the manure that is deposited annually on soils by livestock on PRP. Estimates of annual manure N from these livestock were derived from animal population and weight statistics; information on the fraction of the total population of each animal type that is on pasture, range, or paddock; and annual N excretion rates for each animal type. The amount of manure N from each animal type was summed over all animal types to yield total PRP manure N. Nitrous oxide emissions resulting from manure deposited on pastures by livestock was simulated by DAYCENT in each county. The emissions were obtained by multiplying DAYCENT emissions (in g N₂O-N m⁻²) by the total reported pasture area for each county, and summing across all counties to achieve a nationwide value. All of the manure accounted for by

DAYCENT was assumed to come from cattle because DAYCENT has been parameterized to simulate cattle manure, and cattle are responsible for approximately 90 percent of total PRP manure. The PRP manure N from paddocks and rangeland not accounted for by DAYCENT in the pasture component was multiplied by the IPCC default emission factor to estimate N₂O emissions from paddock and rangeland manure deposition. Emissions from the three types of PRP manure were summed to provide total national emissions from PRP manure in the United States.

Indirect N₂O Emissions from Managed Soils of All Land Use Types

This section describes the method for estimating indirect N₂O emissions from managed soils of all land use types (i.e., cropland, grassland, forest land and settlements). Indirect emissions of N₂O are composed of two parts, which are estimated separately and then summed. These parts are 1) emissions resulting from volatilization of non-N₂O gases (i.e., NO_x and NH₃) from synthetic fertilizer and manure additions to managed soils and from managed manure during storage, treatment and transport that are subsequently deposited onto other areas and eventually emitted to the atmosphere as N₂O, and 2) leaching and runoff of N (in the form of NO₃⁻) from all soils where N additions have been made that is eventually denitrified and emitted as N₂O from a water body. Regardless of the original source or eventual land use type where these indirect N₂O emissions actually occur, all indirect N₂O emissions are accounted for in this section of the Inventory.

A mix of approaches was used to obtain the necessary information required to estimate indirect N₂O emissions. While DAYCENT simulates NO_x and NH₃ volatilization as well as NO₃ leaching/runoff, it does not model their transport or subsequent off-site conversion to N₂O. Therefore, DAYCENT was used to simulate N volatilization and leaching/runoff losses for major crop types. Volatilized and leached/runoff N from non-major crops, settlements and forest lands were obtained by applying the IPCC default fractions to total fertilizer applications to those crops and/or land areas. The volatilization and leaching/runoff components of indirect emissions for PRP manure were obtained by using a combination of DAYCENT generated outputs for manure deposited on pasturelands and applying IPCC defaults to manure deposited on paddocks and rangelands. Manure from managed systems assumed to be volatilized during storage, treatment and transport was included in the indirect emission calculations as well. In contrast to the IPCC approach that has been used in the past, DAYCENT simulations for major crops, where all managed manure is assumed to be applied, do not assume that 100 percent of the N in managed manure is available to be applied to soils. According to data in Kellogg et al. (2000) and Edmonds et al. (2003), more than 50 percent of the N in managed manure is lost to volatilization, spillage and leaching/runoff during storage, treatment and transport. Consequently, manure N applied to soils, based on data from Kellogg et al. (2000) and Edmonds et al. (2003), is subtracted from total managed manure N and assumed to volatilize during storage, treatment, and transport where it is then included in the volatilization component of indirect emissions. Results from this mix of approaches described above were then summed for the appropriate indirect N₂O emission pathway as described below.

Volatilized Indirect Emissions

Volatilized N emissions for settlements, forest lands, PRP manure, major crops, non-major crops, and volatilized managed manure prior to land application were summed. The IPCC default emission factor for indirect N₂O was applied to the total to give total indirect N₂O emissions from N volatilization from soils of all land use types and volatilized managed manure.

Leaching/Runoff Indirect Emissions

The amounts of leached/runoff N from settlements, forest lands, PRP manure, major and non-major crop types were summed and multiplied by the IPCC default emission factor for leached/runoff N.

Total Indirect Emissions from Volatilization and Leaching/Runoff

Total indirect emissions from volatilization and from leaching/runoff were summed to estimate total indirect emissions of N₂O from croplands (Table 6-17).

Uncertainty

The DAYCENT biogeochemical ecosystem model was used to calculate N₂O emissions from major crop types. There are two broad classes of uncertainty in such analyses: that inherent in the activity data and emission factors, and structural uncertainty inherent to the model used to estimate emissions. Consistent with the United States' uncertainty management plan, uncertainty inherent to the DAYCENT model was not quantified as part of the IPCC Tier 1 approach described below.

Three types of approaches were taken for estimating different types of emissions in this chapter: 1) Direct emissions calculated by DAYCENT; 2) Direct emissions not calculated by DAYCENT; and 3) Indirect emissions. Uncertainty was estimated differently for each category.

For direct emissions calculated by DAYCENT (99.3 of the total direct 155.3 Tg CO₂ Eq.), uncertainty in national totals for N inputs and uncertainty in how N application rates change with crop type, year, and agricultural region contribute to total uncertainty in the N application activity data. Total uncertainty in N inputs was estimated at 20 percent (Mosier 2004). Other activity data include climate data, for which uncertainty was estimated to be 19 percent, and soil type, which was estimated to have an uncertainty of 12 percent (Del Grosso 2005a). Their combined uncertainty, according to the sum-of-squares method, is approximately 30.1 percent. To estimate the uncertainty associated with the effective emission factor, DAYCENT outputs were compared with N₂O measurements from various cropped soils over the annual cycle (Del Grosso et al. in press). Through this method, the uncertainty associated with the effective emission factor was estimated at 57 percent (Del Grosso 2005b). Through the calculus of error propagation, overall uncertainty for direct emissions calculated by DAYCENT was 64 percent.

Direct N₂O emissions not calculated by DAYCENT were assumed to maintain the 64 percent uncertainty.

Finally, indirect emissions were calculated according to the default IPCC methodology, as has been performed in past Inventories. Consequently, the maximum uncertainty calculated for last year's indirect N₂O emissions from agricultural soil management of 286 percent (U.S. EPA 2004) was applied to conservatively address the uncertainty in indirect emissions here.

The results of the Tier 1 quantitative uncertainty analysis are summarized in Table 6-18. Agricultural soil management N₂O emissions in 2003 were estimated to be between 45.2 and 461.8 Tg CO₂ Eq. at a 95 percent confidence level. This indicates a range of 82 percent above and below the 2003 emission estimate of 253.5 Tg CO₂ Eq.

Table 6-18: Tier 1 Quantitative Uncertainty Estimates of N₂O Emissions from Agricultural Soil Management in 2003 (Tg CO₂ Eq. and Percent)

Source	Gas	2003 Emission Estimate (Tg CO ₂ Eq.)	Uncertainty (%)	Uncertainty Range Relative to Emission Estimate (Tg CO ₂ Eq.)	
				Lower Bound	Upper Bound
Agricultural Soil Management	N ₂ O	253.5	82%	45.2	461.8

Recalculations Discussion

Differences in the present report compared to previous years exist for two reasons: differences in sources and differences in methodologies. In previous Inventories, fertilizer applied to forests and settlements were included in the agricultural sector. For the current Inventory, for the direct emissions, these fertilizer additions were included in the LUCF sector, and therefore approximately 15 percent less synthetic fertilizer is counted in the agricultural sector than in previous Inventories. Also in previous Inventories, the default Tier 1 IPCC methodology was used to estimate emissions from this sector. That methodology relied solely on N inputs, and did not account for effects of climate, soil type, and other factors that influence N₂O emissions. To account for some of these additional factors and increase confidence in estimates, a Tier 3 method, the DAYCENT ecosystem model, was used to account for

N₂O emissions from major cropping systems. Overall, the changes resulted in an average annual decrease of 31.2 Tg CO₂ Eq. (11 percent) in N₂O emissions from agricultural soil management for the period 1990 through 2002.

The IPCC emission factor methodology is an example of a Tier 1 approach. This approach is activity driven, i.e., total N from different sources (e.g. synthetic fertilizer, manure, N fixation, etc.) is used to estimate N₂O from these sources. The Tier 3 approach in this case uses a process-based model (i.e., DAYCENT) and is area driven, i.e., it is necessary to know the annual area of major crop types and the N amendment rates for each of these crops. With the Tier 3 approach, emissions cannot be separated by N inputs because once N is in the plant/soil system, the model does not distinguish its source according to IPCC categorizations (e.g., whether the N₂O emitted was synthetic fertilizer-derived or derived from manure). Because the Tier 3 approach was used for approximately 90 percent of fertilized soils in the United States, N₂O emissions are not partitioned into the IPCC's N-input categories, as has been done in the past.

The Tier 3 approach requires some of the same activity data as the Tier 1 approach, plus additional information. Like the Tier 1 approach, the Tier 3 approach requires national totals for N amendments, but it also requires data on N amendment rates for different cropping systems. Consequently, the total amounts of N fertilizer and organic N additions were identical to previous years but assumptions regarding the fate of these amendments are different. For example, in previous years, 100 percent of managed manure was assumed to be applied to cropped soils, though here approximately 64 percent of manure N was lost to volatilization during transport and storage before it was applied to soil. This manure that was assumed to volatilize before soil application was included with indirect emissions, which is different than previous years. In addition to N amendments, the Tier 3 approach requires area data for different cropping systems. The Tier 3 approach distinguishes different cropping systems because crops vary in growth rates, fertilization rates, biomass N concentration, and timing for planting, harvesting, and cultivating. These crop system specific factors are important because they influence N availability in soil, which controls N₂O emissions.

An important difference between Tier 1 and Tier 3 approaches relates to assumptions regarding N cycling. Tier 1 assumes that N added to a system in one year completely cycles during that year; e.g., N added as fertilizer or through fixation contributes to N₂O emission for that year, but cannot be stored in soil or biomass and be recycled and contribute to N₂O emission in subsequent years. In contrast, the process-based models used in the Tier 3 approach include legacy effects such that N added to the system in one year may be taken up by vegetation and returned to the soil in organic form during that year, then re-mineralized and emitted as N₂O during subsequent years. In addition to previous years' fertilizer additions, other long-term management practices that affect current soil organic matter (SOM) levels (e.g., intensive cultivation, summer fallow) also affect current N₂O emission, because in process based models, N from internal cycling (mineralization of SOM) contributes to N₂O emission. Thus, while Tier 1 estimates are influenced only by the current year's N inputs, Tier 3 emissions are also influenced by management in previous years.

Another difference in methodologies is that the Tier 1 method assumes that 10 percent of synthetic fertilizer and 20 percent of applied manure are volatilized, and 30 percent of applied N is leached or run-off. DAYCENT, however, calculates N volatilization and N leached and run-off internally based on specific climatic, environmental, and management conditions.

Consideration of N-fixation highlights another difference in the approaches. In the Tier 1 approach, a certain portion of aboveground fixed N is assumed to be emitted as direct soil N₂O. In the Tier 3 approach, N fixation is calculated by the model and fixed N can be harvested, lost as N₂O, lost in some other form (e.g., leached NO₃), or stored in the plant/soil system.

The Tier 1 approach also assumes that only N from fertilizer and organic matter additions contributes to indirect N₂O emissions whereas the Tier 3 approach assumes that once N is in the plant/soil system, it can be cycled and lost through various pathways, regardless of its source. Similar to N fixation, N deposited on soil by pasture and range animals and N added to soils from crop residue are simulated by DAYCENT. More N from manure was assumed to volatilize before application to soils and hence less N from manure was available for leaching than previous years. However, total N volatilization and leaching/runoff were both still higher than previous years. This is because IPCC

methodology considers only N from synthetic and organic fertilizer to contribute to indirect emissions whereas other sources of N (e.g., fixation, crop residue) contribute to volatilization and N leaching/runoff in DAYCENT.

The methodology used here estimated total N₂O emissions to be approximately 5 to 10 percent less than estimates based on the IPCC methodology due to changes in the calculation method, as well as accounting for N₂O from fertilization of forest and settlement soils within the LUCF sector. The current method estimates lower direct N₂O emissions Table 6-19, but higher indirect N₂O emissions (Table 6-20) than the IPCC method. Differences in total N₂O emissions are shown in Table 6-21. Direct emissions were lower because of different assumptions regarding the cycling of fixed N and lower manure N applications to the major crop types under the current methodology compared with that used in the past. Indirect emissions, on the other hand, were larger because more contributors to N volatilization and leaching/runoff are accommodated by the simulation (by including crop residue applications, for example). Mean direct emissions from non-N fixing crops differed by approximately one percent, whereas direct emissions from N fixing crops were approximately 30 percent less with hybrid than IPCC methodology. Interestingly, total N fixation with the hybrid approach was only approximately two percent lower than with IPCC methodology and the implied emission factor for direct N₂O emissions from fixation is approximately 0.9 percent using hybrid methodology; i.e., these DAYCENT simulations suggest that the 1.25 percent emissions factor used for direct N₂O emissions from N fixation is too high. This is consistent with field data showing that IPCC methodology may overestimate N₂O emissions from soybean and alfalfa cropping (Del Grosso et al. in press, Rochette et al. 2004).

Table 6-19. Comparison of Direct Soil N₂O Emission Estimates for IPCC versus Current Methodologies (Tg CO₂ Eq.).

Method	1990	1997	1998	1999	2000	2001	2002	2003
IPCC	191.2	214.9	216.1	213.9	213.0	213.2	210.1	205.8
Current Simulation*	146.0	162.3	165.1	157.7	162.6	160.7	166.3	161.7
Difference	45.2	52.6	51.0	56.2	50.4	52.5	43.8	44.1

* Unlike Table 6-13, emissions due to N applied to forest lands and settlements are included here, to be consistent with IPCC estimates used in previous reports.

Table 6-20. Comparison of Indirect Soil N₂O Emission Estimates for IPCC versus Current Methodologies (Tg CO₂ Eq.).

Method	1990	1997	1998	1999	2000	2001	2002	2003
IPCC	72.6	79.0	78.8	78.8	77.4	76.0	77.2	77.3
Current Simulation	112.6	96.2	109.1	92.3	107.6	102.6	92.7	98.2
Difference	-40.0	-17.2	-30.3	-13.5	-30.2	-26.6	-15.5	-20.9

Table 6-21. Comparison of Total Soil N₂O Emission Estimates for IPCC versus Current Methodologies (Tg CO₂ Eq.).

Method	1990	1997	1998	1999	2000	2001	2002	2003
IPCC	263.8	293.9	294.9	292.8	290.4	289.2	287.2	283.1
Current Simulation*	258.6	258.4	274.2	250.0	270.2	263.3	259.0	259.9
Difference	5.2	35.4	20.7	42.7	20.2	25.9	28.2	23.2

* Unlike Table 6-13, emissions due to N applied to forest land and settlements are included here, to be consistent with IPCC estimates used in previous reports.

Compared with the IPCC methodology used in the past, the current methodology shows a smaller increase in total N₂O emissions from 1990 through 2003. The current methodology takes into account climate patterns as well as annual fluctuations in N additions. The linear regression between emissions estimated with the new method and time shows a trend toward increasing emissions of approximately 0.39 percent per year. During this time period, synthetic N fertilizer applications increased by nine percent, manure additions increased by 11 percent, and N fixation increased by about 17 percent. Soybean cropped area increased by 27 percent, corn area increased by six percent, and wheat area decreased by 20 percent. The increase in soybean area is largely responsible for the increase in fixation. Because total non-legume cropped area decreased and total fertilizer applied to major crops

increased, the average rate of fertilizer applied to major crops increased by 32 percent from 1990 through 2003. The current method accounts for each of these variables plus the effects of climate variability, whereas the previous method accounted only for changes in fertilizer and manure additions. Climate interacts with N additions to control emissions with the new methodology. Total N additions from fertilizer are important with the IPCC methodology, while the current method accounts for total N additions, the area that receives the N are important, as well as environmental and management conditions. As a result, simulated N₂O emission estimates may increase or decrease non-linearly, whereas emissions always increase linearly with N applications when using the IPCC methodology.

Planned Improvements

The presented uncertainty estimate is incomplete in that uncertainty in model activity data besides N inputs (county level weather and soil type) was not included. Because county level soil and climate data are applied across the entire county, within which a great deal of variability may occur, there is inherent uncertainty in assuming that soil type and climate do not vary within a county. Future estimates of uncertainty will include sensitivity analyses so that the response of model N₂O output to variations in climate, soil type, and N inputs can be quantified. Also, a more appropriate methodology than Tier 1 will also be used in future uncertainty estimates. Future efforts at characterizing uncertainty will work toward the inclusion of all agricultural soil management subsource categories in a Monte Carlo style calculation.

6.5. Field Burning of Agricultural Residues (IPCC Source Category 4F)

Large quantities of agricultural crop residues are produced by farming activities. A variety of ways exist to dispose of these residues. For example, agricultural residues can be left on or plowed back into the field, composted and then applied to soils, landfilled, or burned in the field. Alternatively, they can be collected and used as fuel, animal bedding material, or supplemental animal feed. Field burning of crop residues is not considered a net source of CO₂, because the carbon released to the atmosphere as CO₂ during burning is assumed to be reabsorbed during the next growing season. Crop residue burning is, however, a net source of CH₄, N₂O, CO, and NO_x, which are released during combustion.

Field burning is not a common method of agricultural residue disposal in the United States; therefore, emissions from this source are minor. The primary crop types whose residues are typically burned in the United States are wheat, rice, sugarcane, corn, barley, soybeans, and peanuts. Of these residues, less than 5 percent is burned each year, except for rice.⁸ Annual emissions from this source over the period 1990 through 2003 have remained relatively constant, averaging approximately 0.7 Tg CO₂ Eq. (35 Gg) of CH₄, 0.4 Tg CO₂ Eq. (1 Gg) of N₂O, 737 Gg of CO, and 32 Gg of NO_x (see Table 6-22 and Table 6-23).

Table 6-22: Emissions from Field Burning of Agricultural Residues (Tg CO₂ Eq.)

Gas/Crop Type	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	0.7	0.8	0.8	0.8	0.8	0.8	0.7	0.8
Wheat	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Rice	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Sugarcane	+	+	+	+	+	+	+	+
Corn	0.3	0.3	0.3	0.3	0.4	0.3	0.3	0.4
Barley	+	+	+	+	+	+	+	+
Soybeans	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Peanuts	+	+	+	+	+	+	+	+
N₂O	0.4	0.4	0.5	0.4	0.5	0.5	0.4	0.4
Wheat	+	+	+	+	+	+	+	+

⁸ The fraction of rice straw burned each year is significantly higher than that for other crops (see “Methodology” discussion below).

Rice	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+
Corn	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Barley	+	+	+	+	+	+	+	+
Soybeans	0.2	0.3	0.3	0.3	0.3	0.3	0.3	0.2
Peanuts	+	+	+	+	+	+	+	+
Total	1.1	1.2	1.2	1.2	1.2	1.2	1.1	1.2

+ Does not exceed 0.05 Tg CO₂ Eq.

Note: Totals may not sum due to independent rounding.

Table 6-23: Emissions from Field Burning of Agricultural Residues (Gg)*

Gas/Crop Type	1990	1997	1998	1999	2000	2001	2002	2003
CH₄	33	37	38	37	38	37	34	38
Wheat	7	6	6	5	5	5	4	6
Rice	4	3	3	4	4	4	3	5
Sugarcane	1	1	1	1	1	1	1	1
Corn	13	16	17	16	17	16	15	17
Barley	1	1	1	+	1	+	+	0
Soybeans	7	10	10	10	10	11	10	9
Peanuts	+	+	+	+	+	+	+	0
N₂O	1	1	1	1	1	1	1	1
Wheat	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+
Corn	+	+	+	+	+	+	+	+
Barley	+	+	+	+	+	+	+	+
Soybeans	1	1	1	1	1	1	1	1
Peanuts	+	+	+	+	+	+	+	+
CO	689	767	789	767	790	770	706	794
Wheat	137	124	128	115	112	98	81	117
Rice	86	72	65	76	76	77	60	96
Sugarcane	18	21	22	23	24	23	23	23
Corn	282	328	347	336	353	338	320	360
Barley	16	13	13	10	12	9	8	10
Soybeans	148	207	211	204	212	222	211	186
Peanuts	2	2	2	2	2	3	2	3
NO_x	28	34	35	34	35	35	33	33
Wheat	4	3	3	3	3	3	2	3
Rice	3	3	2	3	3	3	2	3
Sugarcane	+	+	+	+	+	+	+	+
Corn	7	8	8	8	8	8	8	9
Barley	1	+	+	+	+	+	+	+
Soybeans	14	20	20	19	20	21	20	18
Peanuts	+	+	+	+	+	+	+	+

* Full molecular weight basis.

+ Does not exceed 0.5 Gg

Note: Totals may not sum due to independent rounding.

Methodology

The methodology for estimating greenhouse gas emissions from field burning of agricultural residues is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997)⁹. In order to estimate the amounts of carbon and nitrogen released during burning, the following equations were used:¹⁰

$$\begin{aligned} \text{Carbon Released} = & (\text{Annual Crop Production}) \times (\text{Residue/Crop Product Ratio}) \\ & \times (\text{Fraction of Residues Burned in situ}) \times (\text{Dry Matter Content of the Residue}) \\ & \times (\text{Burning Efficiency}) \times (\text{Carbon Content of the Residue}) \times (\text{Combustion Efficiency})^{11} \end{aligned}$$

$$\begin{aligned} \text{Nitrogen Released} = & (\text{Annual Crop Production}) \times (\text{Residue/Crop Product Ratio}) \\ & \times (\text{Fraction of Residues Burned in situ}) \times (\text{Dry Matter Content of the Residue}) \\ & \times (\text{Burning Efficiency}) \times (\text{Nitrogen Content of the Residue}) \times (\text{Combustion Efficiency}) \end{aligned}$$

Emissions of CH₄ and CO were calculated by multiplying the amount of carbon released by the appropriate IPCC default emission ratio (i.e., CH₄-C/C or CO-C/C). Similarly, N₂O and NO_x emissions were calculated by multiplying the amount of nitrogen released by the appropriate IPCC default emission ratio (i.e., N₂O-N/N or NO_x-N/N).

The crop residues that are burned in the United States were determined from various state-level greenhouse gas emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992).

Crop production data for all crops except rice in Florida and Oklahoma were taken from the USDA's *Field Crops, Final Estimates 1987-1992, 1992-1997* (USDA 1994, 1998), *Crop Production 1999 Summary* (USDA 2000), *Crop Production 2000 Summary* (USDA 2001), *Crop Production 2001 Summary* (USDA 2002), *Crop Production 2002 Summary* (USDA 2003) and *Crop Production 2003 Summary* (USDA 2004). Rice production data for Florida and Oklahoma, which are not collected by USDA, were estimated by applying average primary and ratoon crop yields for Florida (Schueneman and Deren 2002) to Florida acreages (Schueneman 1999b, 2001; Deren 2002; Kirstein 2003, 2004; Cantens 2004) and Oklahoma acreages¹² (Lee 2003, 2004). The production data for the crop types whose residues are burned are presented in Table 6-24.

The percentage of crop residue burned was assumed to be 3 percent for all crops in all years, except rice, based on state inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, and Cibrowski 1996). Estimates of the percentage of rice residue burned were derived from state-level estimates of the percentage of rice area burned each year, which were multiplied by state-level, annual rice production statistics. The annual percentages of rice area burned in each state were obtained from the agricultural extension agents in each state and reports of the California Air Resources Board (CARB) (Bollich 2000; Deren 2002; Guethle 1999, 2000, 2001, 2002, 2003, 2004; Fife 1999; California Air Resources Board 1999, 2001;

⁹ The IPCC Good Practice Guidance (IPCC 2000) provided no updates the methodology for estimating field burning of agricultural residues.

¹⁰ Note: As is explained later in this section, the fraction of rice residues burned varies among states, so these equations were applied at the state level for rice. These equations were applied at the national level for all other crop types.

¹¹ Burning Efficiency is defined as the fraction of dry biomass exposed to burning that actually burns. Combustion Efficiency is defined as the fraction of carbon in the fire that is oxidized completely to CO₂. In the methodology recommended by the IPCC, the "burning efficiency" is assumed to be contained in the "fraction of residues burned" factor. However, the number used here to estimate the "fraction of residues burned" does not account for the fraction of exposed residue that does not burn. Therefore, a "burning efficiency factor" was added to the calculations.

¹² Rice production yield data are not available for Oklahoma so the Florida values are used as a proxy.

Klosterboer 1999a, 1999b, 2000, 2001, 2002, 2003; Lindberg 2002, 2003, 2004; Linscombe 1999a, 1999b, 2001, 2002, 2003, 2004; Mutters 2002, 2003; Najita 2000, 2001; Schueneman 1999a, 1999b, 2001; Slaton 1999a, 1999b, 2000; Stansel 2004; Street 1999a, 1999b, 2000, 2001, 2002, 2003; Walker 2004; Wilson 2001, 2002, 2003, 2004) (see Table 6-25 and Table 6-26). The estimates provided for Florida remained constant over the entire 1990 through 2003 period, while the estimates for all other states varied over the time series. For California, the annual percents of rice area burned in the Sacramento Valley are assumed to be representative of burning in the entire state, because the Sacramento Valley accounts for over 95 percent of the rice acreage in California (Fife 1999). These values declined between 1990 and 2003 because of a legislated reduction in rice straw burning (Lindberg 2002) (see Table 6-26).

All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stütze (1987). The datum for sugarcane is from University of California (1977). Residue dry matter contents for all crops except soybeans and peanuts were obtained from Turn et al. (1997). Soybean dry matter content was obtained from Strehler and Stütze (1987). Peanut dry matter content was obtained through personal communications with Jen Ketzis (1999), who accessed Cornell University's Department of Animal Science's computer model, Cornell Net Carbohydrate and Protein System. The residue carbon contents and nitrogen contents for all crops except soybeans and peanuts are from Turn et al. (1997). The residue carbon content for soybeans and peanuts is the IPCC default (IPCC/UNEP/OECD/IEA 1997). The nitrogen content of soybeans is from Barnard and Kristoferson (1985). The nitrogen content of peanuts is from Ketzis (1999). These data are listed in Table 6-27. The burning efficiency was assumed to be 93 percent, and the combustion efficiency was assumed to be 88 percent, for all crop types (EPA 1994). Emission ratios for all gases (see Table 6-28) were taken from the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Table 6-24: Agricultural Crop Production (Gg of Product)

Crop	1990	1997	1998	1999	2000	2001	2002	2003
Wheat	74,292	67,534	69,327	62,569	60,758	53,262	43,992	63,590
Rice	7,113	8,346	8,578	9,391	8,703	9,794	9,601	9,050
Sugarcane	25,525	28,766	30,896	32,023	32,762	31,377	32,597	31,178
Corn*	201,534	233,864	247,882	239,549	251,854	241,485	228,805	256,905
Barley	9,192	7,835	7,667	6,103	6,939	5,430	4,940	6,011
Soybeans	52,416	73,176	74,598	72,223	75,055	78,671	74,291	65,795
Peanuts	1,635	1,605	1,798	1,737	1,481	1,940	1,506	1,880

*Corn for grain (i.e., excludes corn for silage).

Table 6-25: Percentage of Rice Area Burned by State

State	Percent Burned 1990-1998	Percent Burned 1999	Percent Burned 2000	Percent Burned 2001	Percent Burned 2002	Percent Burned 2003
Arkansas	13	13	13	13	16	22
California	variable ^a	27	27	23	13	14
Florida ^b	0	0	0	0	0	0
Louisiana	6	0	5	4	3	3
Mississippi	10	40	40	40	8	65
Missouri	5	5	8	5	5	4
Oklahoma	3	3	3	3	3	0
Texas	1	2	0	0	0	0

^a Values provided in Table 6-26.

^b Although rice is cultivated in Florida, crop residue burning is illegal. Therefore, emissions remain 0 throughout the time series.

Table 6-26: Percentage of Rice Area Burned in California

Year	California
1990	75
1991	75
1992	66
1993	60

1994	69
1995	59
1996	63
1997	34
1998	33

Table 6-27: Key Assumptions for Estimating Emissions from Field Burning of Agricultural Residues

Crop	Residue/Crop Ratio	Fraction of Residue Burned	Dry Matter Fraction	Carbon Fraction	Nitrogen Fraction	Burning Efficiency	Combustion Efficiency
Wheat	1.3	0.03	0.93	0.4428	0.0062	0.93	0.88
Rice	1.4	variable	0.91	0.3806	0.0072	0.93	0.88
Sugarcane	0.8	0.03	0.62	0.4235	0.0040	0.93	0.88
Corn	1.0	0.03	0.91	0.4478	0.0058	0.93	0.88
Barley	1.2	0.03	0.93	0.4485	0.0077	0.93	0.88
Soybeans	2.1	0.03	0.87	0.4500	0.0230	0.93	0.88
Peanuts	1.0	0.03	0.86	0.4500	0.0106	0.93	0.88

Table 6-28: Greenhouse Gas Emission Ratios

Gas	Emission Ratio
CH ₄ ^a	0.005
CO ₂ ^a	0.060
N ₂ O ^b	0.007
NO _x ^b	0.121

^a Mass of carbon compound released (units of C) relative to mass of total carbon released from burning (units of C).

^b Mass of nitrogen compound released (units of N) relative to mass of total nitrogen released from burning (units of N).

Uncertainty

One source of uncertainty in the calculation of non-CO₂ emissions from field burning of agricultural residues is in the estimates of the fraction of residue of each crop type burned each year. Data on the fraction burned, as well as the gross amount of residue burned each year, are not collected at either the national or state level. In addition, burning practices are highly variable among crops, as well as among states. The fractions of residue burned used in these calculations were based upon information collected by state agencies and in published literature. Based on expert judgment, uncertainty in the fraction of crop residue burned ranged from zero to 100 percent, depending on the state and crop type.

Based on expert judgment, the uncertainty in production for all crops considered here is estimated to be 5 percent.

Residue/crop product ratios can vary among cultivars. For all crops except sugarcane, generic residue/crop product ratios, rather than ratios specific to the United States, have been used. An uncertainty of 10 percent was applied to the residue/crop product ratios for all crops.

Based on the range given for measurements of soybean dry matter fraction (Strehler and Stützel 1994), residue dry matter contents were assigned an uncertainty of 3.1 percent for all crop types.

Burning and combustion efficiencies were assigned an uncertainty of 5 percent based on expert judgment.

The N₂O emission ratio was estimated to have an uncertainty of 28.6 percent based on the range reported in IPCC (2000). The uncertainty estimated for the CH₄ emission ratio was 40 percent based on the range of ratios reported in IPCC (2000).

The results of the Tier 1 quantitative uncertainty analysis are summarized in Table 6-29. Field burning of agricultural residues CH₄ emissions in 2003 were estimated to be between 0.2 and 1.3 Tg CO₂ Eq. at a 95 percent

confidence level. This indicates a range of 69 percent above and below the 2003 emission estimate of 0.8 Tg CO₂ Eq. Also at the 95 percent confidence level, N₂O emissions were estimated to between 0.1 and 0.7 Tg CO₂ Eq. (or approximately 68 percent above and below the 2003 emission estimate of 0.4 Tg CO₂ Eq.).

Table 6-29: Tier 1 Quantitative Uncertainty Estimates for CH₄ and N₂O Emissions from Field Burning of Agricultural Residues (Tg CO₂ Eq. and Percent)

Agricultural Residues (Tg CO ₂ Eq. and Percent)					
Source	Gas	2003	Uncertainty	Uncertainty Range Relative to	
		Emission		2003 Emission Estimate	
		Estimate		(Tg CO ₂ Eq.)	
		(Tg CO ₂ Eq.)	(%)	Lower Bound	Upper Bound
Field Burning of Agricultural Residues	CH ₄	0.8	69%	0.2	1.3
Field Burning of Agricultural Residues	N ₂ O	0.4	68%	0.1	0.7

Recalculations Discussion

For the current Inventory, a transcription error was fixed for the 1998 rice production data for California from the USDA 2000 Crop Production Summary Report (2001). The change resulted in increases of less than 0.1 Tg CO₂ Eq. (0.1 percent) in CH₄ and N₂O emissions from the field burning of agricultural residues for 1998. Additionally, the 2002 rice production data was updated from the USDA 2003 Crop Production Summary Report (2004). The change resulted in increases of less than 0.1 Tg CO₂ Eq. (0.2 and 0.4 percent, respectively) in CH₄ and N₂O emissions from the field burning of agricultural residues for that year.

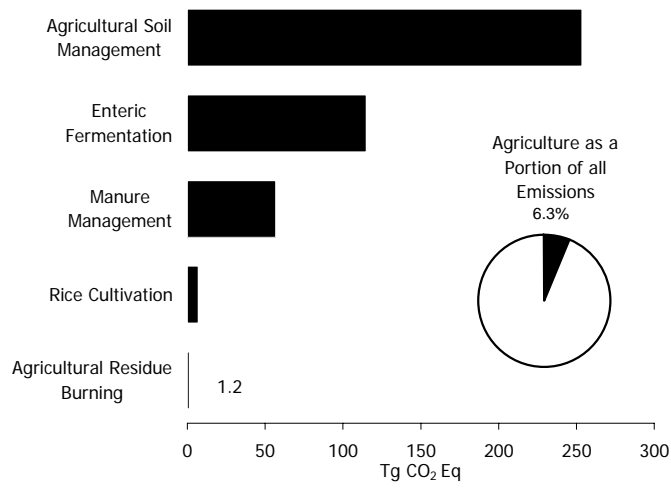


Figure 6-1: 2003 Agriculture Chapter GHG Sources

Figure 6-2

